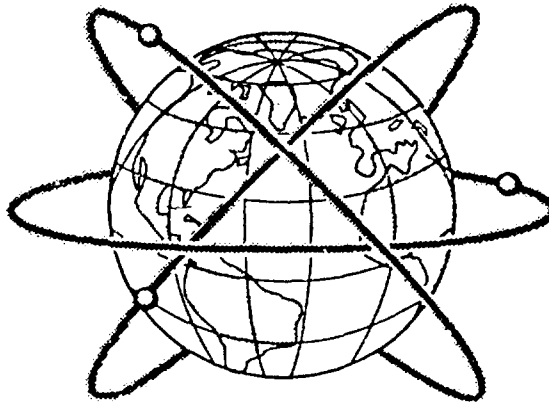


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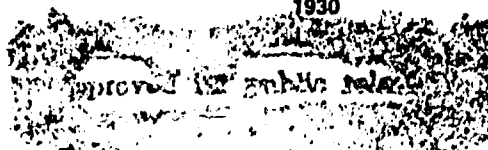
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*Radioactivity and Environmental Security in the Oceans:  
New Research and Policy Priorities in the Arctic and North Atlantic*

*June 7-9, 1993*

*Woods Hole Oceanographic Institution  
Woods Hole, Massachusetts USA*



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**Erratum:**

**Page 481, in the second full paragraph, line 15 and 16 should read  
"emitters with half-lives of at least one half a year; and tritium and  
beta/gamma emitters with half lives of less than one year."**

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# RADIOACTIVITY AND ENVIRONMENTAL SECURITY IN THE OCEANS: NEW RESEARCH AND POLICY PRIORITIES IN THE ARCTIC AND NORTH ATLANTIC

Proceedings of an International Conference  
Held June 7-9, 1993  
At the Woods Hole Oceanographic Institution,  
Woods Hole, MA, USA

## I. Conference Background

### Rationale

Since the mid-1940s, artificial (as distinct from naturally-occurring) radioactive material has made its way into the world's oceans. Much of this material has derived from military activity associated with the Cold War and with accidents at nuclear power plants and other sources. Responsible international management of the resulting environmental threat has been impeded by the strained relations and high level of secrecy produced by that conflict.

With the ending of the Cold War, an opportunity has been created to tackle this problem cooperatively. Furthermore, with the lifting of the veil of secrecy that has hung over the nuclear waste practices of the nuclear weapon states, and indeed over the military nuclear activities of other nuclear weapon states as well, information has begun to emerge that suggests new possibilities for the investigation of this problem on a global basis.

The disposal of radioactive waste at sea has been practiced by many countries, particularly the nuclear weapon states, and dump sites are to be found in many parts of the ocean. Fallout from the atmospheric nuclear testing carried out in the 1940s, '50s and '60s, and the accidental sinking of several nuclear-powered submarines (some carrying nuclear weapons), have also contributed to the global inventory of artificial radioactive materials in the ocean.

The International Atomic Energy Agency has endeavored to maintain a systematic record of this inventory. Casual or negligent dumping practices in the early years of the nuclear age by Western countries, together with the stricter government secrecy of Soviet dumping, have left significant gaps in the IAEA's record, however, gaps that now should be filled.

In addition to questions about what artificial radioactive materials are now in the ocean and where, there also exists considerable debate in scientific and policy circles, and among the public, about the current and potential consequences of the presence of these materials in the ocean on plants, animals and people.

At one end of the spectrum, some contend that any artificial radioactive material in the ocean is unacceptable. At the other, some regard the consequences as negligible.

With the opening up of the issue, an opportunity has emerged to examine the consequences on a systematic, scientific basis. Furthermore, the political and economic pressures to do so are mounting. Concern over the effects of radioactive materials in the ocean on fish has already had serious consequences on the fishing industries of several countries. Further, since radioactive materials in the ocean do not respect national borders, and the waste dumped by one country may enter the waters of another, this issue has incited international tensions and has considerable potential to produce conflicts. Finally, general international concern for the environment has made ocean dumping of any kind, radioactive or not, a major issue for governments and legislators around the world.

Recent articles concerning nuclear dumping in the Arctic, including front-page newspaper stories, have turned this issue into a potential policy time-bomb. There is a sense in many regions of the world that we may so far have glimpsed no more than the tip of the iceberg.

Finally, the existing inventory of radioactive materials in the ocean is not the only problem. Both the United States and Russia, and other nuclear weapon states, have a large number of nuclear-powered submarines approaching obsolescence. In addition there are scores of vessels already laid up and awaiting disposal. As a result, there is a disposal problem of nuclear waste of this kind.

The central rationale for the international conference held in Woods Hole, therefore, is that the ending of the Cold War has created both an opportunity and a compelling need for a cooperative examination of the issues raised by the presence of artificial radioactivity in the world's ocean. The opportunity for cooperation and constructive disclosure is truly unprecedented.

These issues are of both a scientific and policy nature. As noted above, the economic and political dimensions of the problem suggest that before very long, policymakers will come under significant commercial and public pressure to address the problem. At the same time, responsible management of the problem demands that policy be supported by rigorous science, and indeed that the two be effectively integrated.

It is clear that a single conference cannot and will not produce a comprehensive solution to the problem. It can, however, expect to advance significantly the state of knowledge about the problem and help to identify gaps in this knowledge and priorities in scientific research and policy. In particular, the conference was designed to develop a practical agenda, along with a sense of priorities, for both the research and policy dimensions of the problem.

#### Initial Planning

The initial planning for the Conference began at a meeting held in Woods Hole in July of 1992. The meeting was attended by scientists from Russia, Canada, and the USA as well as a number of representatives of government agencies in the USA and Russia (attendees are listed in Appendix I). The participants at this meeting discussed the utility of a Conference whose purpose was to bring together the rapidly expanding information on nuclear materials in the oceans and consider the implications of this information regarding future research and policy options and decided to proceed with holding such a conference, and, further, to attempt to organize biennial conferences on specific aspects of the general topic of radioactivity and the environmental security of the oceans. It was decided to focus the Conference discussions on four broad topics as they pertain to artificial radioactivity in the North Atlantic and Arctic Ocean:

1. Inventory
2. Routes, Rates, and Reactions
3. Disposal and Remediation
4. Legal, Economic and Policy Priorities.

The third topic was later amended to Assessment and Remediation. It was decided that the Conference would be convened by non-governmental institutes from four countries: The Woods Hole Oceanographic Institution (USA); The Canadian Centre for Global Security (Canada); Institutes of the Russian Academy of Sciences (Russia); and the Fridtjof Nansen Institute (Norway). A Planning Committee (Appendix I) was established to develop the topics to be presented at the conference, to identify keynote speakers and participants, and to set the format of the meeting.

#### Objectives

The purpose of the Conference was to re-evaluate, with a free and open exchange of scientific information, the current and future potential impact of artificial radioactivity in the marine environment. Specifically, to assess current scientific knowledge and policy priorities and to recommend new research strategies for the Arctic and North Atlantic.

## **II. Meeting Format, Program, and Participants.**

#### Meeting Format

The meeting period of three days was divided into 6 morning and afternoon sessions. One session was devoted to each of the four themes of the Conference (Inventory; Routes, Rates and Reactions; Assessment and Remediation; Legal, Economic and Policy Priorities). Each of these sessions



included one or two keynote talks, an introduction, by the authors, of the poster presentations of the session theme, and the poster presentations themselves. All posters were available for the duration of the meeting. This format of summary type presentations coupled with poster presentations was chosen in an effort to promote as much discussion and information exchange on the part of all participants as possible. Simultaneous translation in Russian or English, as appropriate, was provided at all plenary sessions. Papers based upon the oral and poster presentations were required of all participants.

Participants broke into working groups according to their interests for the fifth session. There were four working groups, organized along the lines of the four themes. The groups were tasked to prepare a report which considered available knowledge and the activities required to define and evaluate problems related to the radioactive contamination in the Arctic and North Atlantic. In particular it was suggested that the working groups review the adequacy of existing knowledge within their theme topic, identify additional information required for confident assessment of environmental human risks, define future directions and priorities, and to make recommendations for reaching these goals. The groups were, however, free to pursue any course they deemed most useful and productive.

The final session was a plenary session devoted to a presentation of the working group reports by the respective chairs, and discussion of these reports before the conference participants as a whole.

#### Conference Program

The Conference was held from Sunday, June 7 to Thursday, June 10, 1993 at the Woods Hole Oceanographic Institution, Woods Hole, MA, USA. The Conference activities included seven keynote papers and ~forty-five poster presentations. The Conference Program is reproduced in Appendix II.

#### Participation

The conference was widely advertised to bring it to the attention of as broad an audience as possible. An announcement was posted on the Oceans Bulletin Board (OMNET Inc.); advertisements were placed in EOS and Nature; and some 600 brochures were mailed to scientists and institutions in the international marine science community. The conference was attended by 117 registered participants from 10 countries; the greatest representation being from Russia and the USA. Attendance and participation was not limited through selection of presentations, although funds for the travel of Russian scientists did limit the number of Russian scientists able to attend.

#### Sponsorship

The conference was made possible by financial support from a number of US Government agencies. These are:

Office of Naval Research; Geo-Acoustics/Arctic Sciences Division  
Department of State; Bureau of Oceans and International Environmental  
and Scientific Affairs

### III. Working Group Reports

The assessment of the present state of affairs as well as recommendations for future research and policy priorities with regard to artificial radioactivity in the North Atlantic and Arctic that was derived from the deliberations of the many scientists attending this conference is embodied in the working group reports. These reports are the outgrowth of discussions by the working group participants as amended to take into account plenary discussions of the reports by the conference participants as a whole.



**Working Group Reports**

**Conference on Radioactivity and Environmental Security in the Oceans: New Research and Policy Priorities in the Arctic and North Atlantic**

**Report of Inventories/Sources Group - Chair; J.N. Smith**

The limited quantity of radionuclide data available from the Russian Arctic indicates that present levels of artificial radionuclides are low compared to other contaminated marine environments such as the Irish Sea. However, the existing radionuclide inventory data set for the Russian marginal seas, the Barents and Kara Seas in particular, is incomplete and the identities and magnitudes of the radionuclide source terms have not yet been fully determined. The Inventories/Sources Group concluded that the following subjects and recommendations should receive priority attention.

1. The existing data base on artificial radionuclide levels and inventories in the Russian Arctic should be consolidated in a form amenable for input to dose assessment models. Russian reports and papers which have received limited circulation should be translated into English. Provision should be made for the translation into Russian of critical papers in English. The major radionuclide sources, the key types of radionuclides and the primary sources of information are given in an accompanying table.

2. Data is lacking on the composition, types of containment and the physico-chemical form of liquid and solid radioactive wastes dumped in the Barents and Kara Seas. It is recommended that data on these subjects be compiled and released by Russian scientists and authorities and that additional, site-specific data be collected during the next few years. Of particular concern are the corrosion rates of waste containment vessels in bottom shelf waters of the Arctic Ocean.

3. Detailed data sets pertaining to the magnitude and composition of radionuclide levels in nuclear reactors dumped on the seabed and the engineering diagnostics of the reactors themselves should be compiled and released by Russian authorities. Site-specific studies of corrosion and degradation of reactor containment systems, including the furfural hardening mixtures used in some instances to isolate reactor cores, should be undertaken.

4. Data on radionuclide transport into the Arctic Ocean from Russian rivers, including the Ob and Yenesie Rivers and smaller systems on the Kola Peninsula, should be compiled and released. The data base should be augmented by site-specific studies which will be undertaken during the next few years. The impact of major, pulsed releases from reactors and waste repositories, or of flood events which could remobilize and transport radionuclides sequestered in catchment basins or holding ponds should be evaluated.

5. Methods for distinguishing between different radionuclide sources (fallout, reprocessing plants, submarine reactors, land-based nuclear reactors) should be applied to studies in the Russian Arctic Ocean. These include tracer techniques (I-129/I-127; Pu-240/Pu-239; U-236/U-235) and in-situ analytical techniques (ie. towed seabed Ge detectors) which are presently under development.

6. At least five oceanographic cruises to the Barents and Kara Seas are to be undertaken during the 1993 field season, yet there is little coordination between the different cruises. Therefore, it is recommended that mechanisms, possibly formulated by a planning committee, be established for collaboration, coordination and sample exchange among the participants in the various cruises to avoid duplication of activities. The efficient dissemination of data from the wide range of ongoing scientific investigations should be promoted by the establishment of international workshops.

7. Protocols should be established, possibly through the auspices of the International Atomic Energy Agency, for the methods and procedures governing the collection and analysis of arctic samples. Laboratories active in these investigations should participate in international intercalibration exercises. Protocols should also be established for the archiving of samples which are being collected on relatively short notice in arctic regions. Samples presently being archived in Russia should be accessed and made available to scientists from other countries.

8. Levels of natural radionuclides (Po-210, Pb-210, Ra-226, thorium decay series, etc.) in different environmental phases in the Arctic Ocean should be evaluated in order to: (1) facilitate their use as tracers to determine water mass transport, particle scavenging rates for anthropogenic radionuclides, radionuclide transport rates through the drainage basin, and (2) provide input data for dose assessment models in order to permit comparisons between natural and artificial radiological assessments.

9. Predictions of future variability in the radioactive source terms should be undertaken including factors such as; a) the further decommissioning of submarines and other nuclear powered vessels, b) future dumping of radioactive wastes and degradation or disturbance of present dumpsites, and c) the possible resumption of nuclear weapons testing at Novaya Zemlya.

10. Following the identification of the primary radioactivity sources in the Russian Arctic, site-specific monitoring programs should be established in order to assess the temporal variability of the source terms. These monitoring programs should provide input data for models evaluating long term environmental effects in addition to the immediate radiological impact of artificial radionuclides on fisheries and human health.

Table 1. Radioactivity sources, the types of inputs, the categories of key radionuclides (actinides, Ac.; fission products, Fis; corrosion products, Corr.; activation products; Act), and the primary sources of data for each input.

Source	Type	Nuclides	Data Source
Coastal	Sellafield	Ac., Fis., Act.	UK
	Cap La Hague	Ac., Fis., Act.	France
	Liquid Waste	Ac., Fis., Act.	Russia
	Floating Storage	Ac., Fis., Act.	Russia
Atomosphere	Fallout/Tests	Ac., Fis., Act.	Russia/Norway
	Chernobyl	Fis. Act.	Russia/Norway
River	Nuclear/ Reactor Waste	Ac., Fis., Act.	Russia
	Non-nuclear Industrial	Fis. Act.	Russia
	Fallout/Tests	Ac., Fis., Act.	Russia
Seabed	Fuelled Reactors	Ac., Fis., Act.	Russia/USA
	Reactor Compartments	Act.	Russia
	Mixed Waste (resins, components, drums)	Act., Corr.	Russia
	Komsomolets	Ac., Fis., Act.	Russia
	Thule (nuclear weapons)	Ac.	Denmark, USA, Canada



**Radioactivity & Environmental Security in the Oceans:  
New Research and Policy Priorities in the Arctic and North Atlantic  
June 7-9, 1993  
Woods Hole, Massachusetts, USA**

**Report of the  
Working Group on  
Routes, Rates, Reactions and Fates**

**Sources and Region of Concern**

The working group delineated the region of concern by defining the Arctic as the area encompassing the maximum extent of sea ice, ice sheets and permafrost, extending south to the Greenland-Faroe sill in the North Atlantic. Past and existing sources of potential leakage of radionuclides into this region include: reactors dumped in Novaya Zemlya fjords, dumpsites in the Novaya Zemlya depression, the Komsomolets submarine, liquid waste released in the Barents Sea, mixed chemical and radioactive waste dumped at various locations, Pacific dumpsites, discharges from Sellafield/La Hague/ Chernobyl, global and local weapons testing fallout, the Ob river, and natural, background radioactivity. In addition, because attention has focused on this issue only recently, we note that there may be other sources that are as yet unidentified.

Potential future sources include: continued dumping of liquid waste in the Barents Sea, leakage from submarines moored near Kola Peninsula, discharges from the Ob and Yenisey rivers, as well as other input from land-based sources, accidents—at sea and on land, resumption of nuclear testing, and other, as yet unknown events.

Because not all present and future releases can be defined, working group participants focused attention on regional and circum-Arctic processes as well as processes in the direct vicinity of known dumpsites.

**Routes and Rates**

Media that may affect transport of pollutants include: water (groundwater, rivers, oceans), sediment (turbidity currents, nepheloid layers, dredging, slumps), ice (permafrost, river ice, sea ice, icebergs), atmosphere (local and regional wind), and biota (transport through food chain, migration).

Regional scales of interest are watersheds, rivers, estuaries, and "airsheds" for input of terrestrial pollutants to the marine environment; local and shelf-scale processes near marine sources; shelf-Arctic Ocean exchange; and general Arctic circulation and transport, including pathways and modes of particle/pollutant transfer.

Transfer and residence times must be defined with attention to natural variability, including the marked seasonality of Arctic processes, interannual variability and assessment of variability during the past 100 years (the period of main anthropogenic influence) as well as the past 15,000 years (the post-glacial period).

A main concern of working group members was assessment of the potential for episodic vs. chronic release of radionuclides. Major events that can cause rapid changes in conditions that could affect pollutant release and/or transfer, include nuclear accidents/major leaks, iceberg/sea ice gouging, tornadoes, storm surges, nuclear testing/venting, high velocity currents/ benthic storms.



## Reactions and Fates

Container/barrier corrosion rates must be assessed with different source material and under various conditions, including attention to such factors as: oxidizing vs. reducing conditions, solubility, organic and colloid content.

Chemistry of released material, including speciation will be a major factor determining eventual fate.

Fate of radionuclides released into the marine environment depends on physical processes, as well as migration and accumulation factors in the food chain.

## Data Required

Working group participants defined six main themes where data is required in order to understand sources, transport and fate of pollutants released into the Arctic marine environment. Data requirements specific to each theme are detailed below. High priority in every case is to define existing information and required new data. In particular, data is required to understand natural variability including temporal and spatial variability, local and large scale effects, biogeochemical processes, and climate change. Analysis of existing data should form the basis for collection of new data.

### Hydrography and local and large-scale circulation

Processes and factors to be assessed include: water mass structure and circulation (currents, and pathways), tides, storms and storm surges, water mass formation and exchange, brine formation and flow along and off shelf, particle transport, bottom boundary layer processes, transient tracers (Sellafield and La Hague), residence and transfer times, ocean-atmosphere interactions.

### Radiochemical speciation and source terms

Processes and factors to be assessed include: input parameterization, source term effects (atmospheric, aquatic), anthropogenic effects, colloidal issues, distribution coefficient, partitioning onto sediment, sorption or desorption, different mixes/different conditions, saturation, migration vs. bioturbation, sediment composition, oxidation states, kinetic data, biological effects.

### Ice dynamics

Processes and factors to be assessed include: distribution and concentration of icebergs, sea, river and estuarine ice, pollutant entrainment and transport by ice, ice drift patterns, off shelf transport, pollutant release from ice, ice-sea/riverbed interactions, gouging, glacial surges, polynyas, internal ice: e.g. effect of permafrost on pollutant transport.

### River and watershed effects

Processes and factors to be assessed include: flow rate, temporal variability, sediment/pollutant storage and delivery, river ice pollutant incorporation and ice gouging, residence times, delay (time from input to effect), groundwater diffusion and migration, estuarine processes: seawater/freshwater interface, hydrologic balances and processes, atmospheric/meteorologic interactions.

### Sediment dynamics

Processes and factors to be assessed include: particle/pollutant associations, particle sources, transport pathways and sinks, terrestrial/fjord/shelf/deep sea particle transport, suspended load transport/nepheloid layers/brine discharge pathways,

major events, cross-shelf sediment transport, sedimentation rates, changes in sediment compositions with time, distribution of sediment types, biological mixing/bioturbation, coastal erosion, catchment areas/geomorphology.

#### Food chain transfer

Processes and factors to be assessed include: transfer to human beings, chemical speciation effects, bioaccumulation: ocean and organisms, pollutant transport by organisms, biota migration, variability—natural and anthropogenic activities, food chain specifics such as diet, especially for indigenous populations, migration, geography, species identification.

#### **Methods and Future Directions**

In order to assess the potential for release of radionuclides into the marine environment as well as to assess understanding of processes affecting pollutant routes, rates, reactions and fates the following activities were identified:

- data exchange
- workshops for data exchange, communication, developing/coordinating future programs
- new measurements including monitoring, time series
- experiments, especially to understand barrier/container corrosion and speciation of pollutants
- development of models for synthesis and prediction

#### **Key Recommendations**

##### Data

Compilation of existing data, including international data exchange/data base/collaboration with emphasis on the Barents and Kara seas, as well as other shelf seas and the central Arctic.

Focus should be on temporal and spatial variability, especially the potential influence of major events to effect pollutant release and transport. Relevant Russian reports must be identified and translated. Classified data relevant to Arctic sources of pollutants and the physical environment, as well as analogue situations should be declassified. Declassification will require political assistance.

##### Data acquisition required:

- pollutant/particle delivery—from the watershed as well as from shelf dumpsites
- pollutant/particle transport and fate—ocean and ice circulation, residence times
- container/barrier corrosion; chemical speciation; sorption/desorption/scavenging; colloidal transport
- food chain processes and effects.

##### Modeling

Development of comprehensive model is required to estimate routes, rates, reactions, fates, to synthesize existing data, and for sensitivity test, assessment of time scales, etc. as a tool for prediction and risk assessment. This will require modeling combined with laboratory and field studies. Modeling should include analysis of episodic processes and development of worst case scenarios.

##### Implementation

In order to define priorities for future efforts, the working group recommended that topic-specific workshops be held to review existing knowledge, identify gaps, prepare a coherent strategy including data acquisition, develop experiments and models, analyze results and communicate findings. When possible, meetings should be held at the relevant Russian institute in order to ferret out expertise and data. Future field efforts should be coordinated to take advantage of existing programs and platforms to gather data on general physical/chemical/biological processes as well as pollutant loading from radionuclides and other pollutants.



## ASSESSMENT AND REMEDIATION

The working group on Assessment and Remediation reviewed the current status of risk assessments pertaining to the major types and locations of known radioactive waste and man-made materials in the North Atlantic and Arctic Ocean. In addition, the threat of potential (future) releases was also considered. Following accepted international practice, the review was based on a table that considered risk in the form of Individual Dose Rate, Collective Dose Commitment, and Effects on Local Biota. Conservative estimates, in most cases guesstimates, of confidence level were discussed and included. Priorities for assessment of the types/locations were assigned, based upon the resulting table. Priority was primarily determined on the basis of either relatively high risk factors or a lack of information on the risk factors. A secondary consideration that affected some assignments of priority was the existence of on-going assessment studies. In such cases, for example the NE Atlantic Dumpsite, a lower priority was assigned as the current program was deemed adequate. Some consideration was given to remediation, but since remediation considerations are, in most cases, either site specific or dependent upon risk factors that are presently unknown or poorly defined, a specific recommendation emerged only in the case of the NE Atlantic Dumpsite where no remediation was deemed indicated or practical.

While some of the sources of radioactive contamination considered in Table 1 are self-explanatory, others need definition. The Deep Point (>1500m) sources discussed were limited to sunken nuclear submarines and their nuclear weapons. The Shallow Arctic concentrated on waste disposal activities in the Kara and Barents Seas. The Shallow Other sources considered refer to lowlevel wastes dumped in shallow water such as those off Massachusetts and the Farallon Islands. Wind-blown dust refers to wind erosion and transport of soil materials. In Potential Future Releases the focus of discussions was upon the very large amounts of radioactivity stored on land in Russia, both wastes from weapons plants and spent fuel, as well as nuclear reactors from decommissioned ships currently stored on shore or afloat in shallow waters of the Arctic coast.

A summary of the priorities recommended by the group is presented in Table 2, and a list of the working group participants given in Table 3.

Table 1: RISK SUMMARY

<u>SOURCE:</u> <u>IMPACT</u> Individual Dose Rate	European Repr. Plants	N.E. Atlantic Dumpsite	Deep Point (1500m)	Shallow Arctic	Shallow Other	Wind-blown Dust	Arctic Rivers	Potential Future Release
	0.1 mSv/yr	10 <sup>-5</sup> mSv/yr	.001mSv/yr	?	0.1 mSv/yr	Probably Minor Effects	?	?
Collective Dose Commitment	5x10 <sup>3</sup> man Sv	4x10 <sup>4</sup> man Sv (C-14)	10 to 100 man Sv (Cs-137)	100 man Sv + Local	?		?	?
Local Biota		Negligible	?	?	?		?	?
Confidence Level	High	Adequate	Factor of 10	Factor of 10	Low			
Verification Priority	Extensive Studies Complete & Continuing	Low (in progress)	Depends on Biota Assessment	High	High ?		High	High
Remediation	In Progress (source reduction)	No	Unlikely, but Depends on Assessment	Evaluate Options	Evaluate Options		Evaluate Options	Evaluate Options

**Table 2: ASSESSMENT PRIORITIES**

### 1. HIGH PRIORITY:

### Shallow Arctic (shelves):

**Primary Considerations:**

- i. individual dose to inhabitants of the region.
- ii. biological Effects.

**Comments:** Oslo Plan supported.

**Shallow (other shelves):**

**Primary Considerations:**

- i. estimated high individual dose.
- ii. lack of detailed dose assessments.

### Arctic Rivers:

**Primary Considerations:**

- i. potential for major release (see next item).

### Potential Future Releases:

**Primary Considerations:**

- i. large inventories**

## 2. LOW PRIORITY

**Deep Point Sources (>1500m; eg sunken weapons, nuclear submarines):**

**Primary Considerations:**

- i. possible local biological effects (expectation is for small or negligible effects; confirmation required)

**Comment:**

**Release of currently classified U.S. data would likely resolve uncertainties.**

### N.E. Atlantic Dumpsite:

**Primary Considerations:**

- i. studies to refine assessment in progress
- ii. Remediation not practical

### European reprocessing plants:

### Primary Considerations:

- i. extensive long term studies completed and ongoing
- ii. Source reduction ongoing



# **POLICY ISSUES AND PRIORITIES**

**SUMMARY REPORT OF THE WORKING GROUP ON  
LEGAL, ECONOMIC AND POLICY PRIORITIES**

by  
**John M. Lamb**  
**Working Group Chair**

**Conference on**  
**Radioactivity and Environmental Security in the Oceans:**  
**New Research and Policy Priorities in the Arctic**  
**and North Atlantic**

**June 7 - 9, 1993**



### **Introduction:**

As noted at the beginning of this *Proceedings*, a central purpose of the Conference was to bring together within a single forum the key scientific and policy issues raised by the past, present and potential future release of artificial radioactive materials into the Arctic and North Atlantic Oceans. Reflected in this approach was a belief that these scientific questions and policy questions are all too often treated as two solitudes. It also reflected a recognition that decisions taken by national governments or international agencies in connection with remediation, regulation and even research will have to be based upon not only available scientific evidence, but also policy considerations relating to legal, political, social and economic matters.

The Conference did not in fact attempt a systematic integration of these scientific and policy issues, recognizing that the questions associated with each area need first to be drawn out and articulated. What appears in this section, therefore, is a presentation of the policy issues discussed during the Conference, together with the specific recommendations put forward by the Conference's Working Group on Legal, Economic and Policy Priorities. Rather than doing so by reporting sequentially on the discussions of policy questions during the plenary, workshop and concluding sessions of the Conference, however, it endeavours to summarize these discussions thematically. Beginning with an outline of the overall context in which policy-makers must take decisions on this issue, the challenges facing them and principles to guide them, it then focuses on the following themes:

- o data and assessment issues
- o social issues
- o economic issues
- o institutional issues
- o legal issues
- o financial issues
- o reducing future inventories of radioactive waste

### **The Policy-Makers' Context:**

As they go about making decisions on these issues, policy-makers must operate in an exceedingly difficult context, including the following:

- o The global inventory of radioactive waste requiring storage and disposal is large and growing.
- o The London Convention (known until 1992 as the London Dumping Convention) in 1972 imposed a prohibition on the disposal of high-level radioactive waste in the oceans. In 1985 LC signatories adopted a resolution placing an indefinite moratorium on all dumping

of radioactive waste at sea. Following the adoption of this 1985 moratorium, an Intergovernmental Panel of Experts on Radioactive Wastes (IGPRAD) was established by the LC to study the issue further, and present options for the future. Its final report, containing seven options ranging from lifting the existing moratorium to establishing a permanent ban on radioactive waste disposal, is to be presented to the LC's 16th Consultative Meeting in November 1993. It is intended that this 16th Consultative Meeting will determine the content of any amendments that are to be made to the LC at the latter's Amendment Conference in the fall of 1994.

- o Public resistance to any ocean disposal of radioactive waste is growing. This may in fact be approaching zero tolerance in some areas - certainly the trend is in that direction.
- o The available scientific evidence relevant to decision-making on the issue remains inadequate, owing in particular to the relatively short periods over which data has been collected, the lack of a coordinated interdisciplinary approach to data assessment, and to the continued classification of government studies and data.
- o For this and other reasons, there is widespread public scepticism or even outright disbelief concerning assurances by the scientific/technical community and political leaders that ocean disposal of radioactive waste can be carried out safely and without prejudice to the ocean environment.
- o Technical options for the storage and disposal of radioactive waste remain under-explored.
- o The legal framework for determining such issues as national and international responsibility and liability relating to radioactive waste is almost non-existent.
- o The financial resources available to governments to tackle these problems are severely limited.
- o Since the late 1980s and especially since the collapse of the Soviet Union in 1991, pressure on Moscow to clean up existing radioactive dump sites in the Arctic Ocean, particularly in the Barrents and Kara Seas, has grown enormously. Since the Soviet Union's dumping practices were exposed in the West in early 1992, international attention on the problem has also expanded greatly.
- o The political power of peoples living in the Arctic is growing, as is their capacity to influence their governments and determine policy outcomes on issues affecting their region.

### **The Policy-Maker's Challenge:**

The continuing growth in quantities of radioactive waste requiring disposal, along with the failure of the LC to impose a permanent ban on its disposal at sea, have meant that the debate over ocean dumping has remained alive. The debate can be expected to intensify until either alternative land-based locations for disposal are agreed upon, or this option is precluded for good by an international ban.

In the meantime, the policy-maker's challenge is to mediate among the various competing pressures listed above in order to determine priorities for the investment of scarce resources in tackling this problem.

In regard to the global problem of radioactive waste, policy-makers face a range of theoretical options, all the way from a general ban on the ocean dumping to a general moratorium, to the easing or lifting of the current moratorium.

Since further intentional disposal of both high- and low-level radioactive waste in the Arctic is already prohibited by the London Convention, there are two major challenges confronting policy-makers in respect to this region: the management and/or remediation of radioactive materials already there; and the prevention of and/or response to potential future accidental releases into the region.

Whatever their scientific merits, the practical impact of scientific assurances that there is no evidence of any regional scale radioactive contamination in the Arctic and the North Atlantic Oceans that currently poses a threat to human health or causes environmental concern is open to question, especially as regards publics and policy-makers. Given the acknowledged need for additional data and studies, these assurances can not be conclusive. In the absence of conclusive evidence that these materials pose no danger, either in the short or long term, these assurances are unlikely to appear persuasive to policy-makers attentive to public constituencies inclined to caution in regard to environmental questions, and to deep-rooted scepticism concerning the advice of scientific experts.

### **The Issue of Principles to Govern Decision-making**

As policy-makers endeavour to meet these challenges, they confront a variety of fundamental questions concerning the principles which should be applied in their decision-making process. These questions reflect contemporary environmental, legal and social uncertainties, as well as perceived public attitudes toward environmental issues. These questions are themselves inherently fluid, often vague, and sometimes contradictory. However, such questions of principle often exercise appreciable influence over government decision-makers.

Among these basic questions are the following:

- o Is society prepared to accept the contamination of certain parts of the Earth, even if that contamination is shown to have negligible direct impact on humankind?
- o should the burden of proof fall on those asserting that there is no significant impact on the environment and humans, or on those asserting that there is such an impact?
- o What time frame should be used in assessing data on these issues. Shouldn't one look beyond the near term impact of radioactivity and consider what happens when the containers break open 50 - 100 years from now?
- o What is an acceptable level of radiation for humans, or for the environment?

While such questions regularly vex decision makers, over the past few years what is known as the "precautionary approach" has come to play a growing part in the international debate over ocean dumping, including endorsement in the Rio Declaration on Environment and Development and in the Earth Summit's Agenda 21. The precautionary approach holds that, given the incomplete state of knowledge about these issues, caution should be the rule and potentially dangerous remedial measures which are irreversible should be avoided.

#### **Data and Assessment Issues**

Perhaps the central point of agreement in the Conference's sessions dealing with policy questions was that additional information is needed in order to permit sound policy decisions. The specific recommendations dealing with data and its assessment were stated as follows:

- o National governments should disclose all existing data, including those currently classified, relating to Arctic environmental contamination.
- o Combined with a comprehensive risk assessment, it is recommended that a systematic analysis be done for a range of options for dealing with high level liquid and solid radioactive waste in the Arctic, and that these options include the design of a cost-effective, comprehensive program for long-term monitoring.
- o National governments should produce and publish an environmental impact statement concerning the operation of nuclear-powered vessels in the Arctic, including an assessment of the operation of these vessels.

- o As part of the systematic analysis proposed above, an evaluation should be conducted of disposal, storage and transport options for high-level radioactive waste of both military and civil origin, anticipated in the future.
- o Scientific and technical information should be gathered to permit comprehensive risk identification, especially of specific sites in the Arctic region, and including the potential risk of transport of radioactive waste by ice.
- o A comprehensive inventory of existing radioactive waste disposal sites in the Arctic should be prepared.
- o National governments should collect, analyze and disseminate data on other land-based source of radioactive waste entering the marine environment.
- o As part of the risk assessment process mentioned above, special attention should be given to the effects of radioactive waste on the long-term health of the peoples living in the Arctic, including contaminants in the food chain.

Further recommendations given appreciable support during the discussions included the following:

- o An assessment is needed concerning the comparative environmental impact of radioactive and other kinds of pollution being released into the Arctic and North Atlantic Oceans.
- o A key gap in data available today concerns the release rates for currently contained radioactive waste situated in the oceans.
- o Further data is needed on food consumption patterns of indigenous peoples around the polar rim.
- o There is a need to develop an inventory of future disposal requirements.
- o Militaries must not be exempted from accountability in regard to the provision of data concerning the dumping of radioactive materials, whether accidental or intentional.
- o There is a need for more information on radioactive release rates from the Russian, U.S. and UK navies.

### **Social Issues:**

8-10 million people live in the Arctic region. As mentioned earlier, these people have in recent years begun to develop a new self-awareness, founded above all on their unique relationship with the natural Arctic environment. This self-awareness has led to growing political power both within their respective countries, and internationally through a variety of regional and global institutions. This political voice has increasingly been turned to the protection of the vulnerable Arctic ecosystem.

This new emerging political context in the circumpolar north suggests that northerners will strongly oppose the disposal of any further nuclear materials in the Arctic Ocean, and will press for a clean-up of existing dump sites.

A recognition of this context led the policy working group to make the following concrete recommendations:

- o The peoples living in the Arctic should be assured of effective and early involvement in the decision-making and implementation processes relating to radioactivity in the Arctic.
- o Consideration should be given to the impact of radioactive waste and remediation options on the subsistence life style and culture of the peoples living in the Arctic.
- o National governments should on a timely basis make available the results of scientific research and monitoring to the peoples living in the Arctic, NGOs, and interested international bodies.

### **Economic Issues:**

As mentioned earlier, the extent of the problems presented by the disposal of radioactive waste and the scarcity of resources demands that policy-makers take hard choices. An appreciation of this fact led the policy working group to make the following recommendation:

- o In view of the limits on public resources, environmentally sound responses to the problem should be as economically efficient as possible, including attention to spill-over effects, hidden subsidies, environmental damages, inter-generational issues, etc., and should recognize that any step taken will divert resources away from other uses.

This led to a discussion of the concept of triage, as practised in emergency medical situations. It was recognized that in determining priorities, a variety of criteria will have to be taken into account.

### **Institutional Issues:**

The London Convention is clearly undergoing a process of transition. With its adoption in 1992 of a 'precautionary approach', there is some likelihood that the current moratorium will be amended to a total ban, or at least a time-limited ban.

However that may be, the trend in international regulation of ocean disposal of radioactive waste has clearly been in the direction of progressively reduced tolerance and increased restriction.

While recognizing the LC as the preeminent global institution governing radioactive waste disposal, certain weaknesses relevant to the Arctic were also recognized, notably its lack of provisions governing accidental release of radioactivity, the lack of provisions governing the disposal of military vessels, and the lack of an accepted definition of government responsibility.

One approach suggested would be to supplement the global regime represented by the London Convention with a regional approach. This would regard the circumpolar Arctic as a coherent region, and build an integrated management regime. It would take into account the unique cultural and environmental nature of the region. It would involve institutional and financial cooperation within the region.

Discussion along these lines led the policy working group to make the following recommendations:

- o Within the framework of the London Convention, special regional measures specific to the Arctic should be addressed through an appropriate regime.
- o The initiatives of the International Arctic Environmental Strategy (Rovaneimi) should be supported and implemented. Enhanced attention should be paid to nuclear contamination.

**Legal Issues:**

In terms of law, neither international nor Arctic regional regimes adequately cover the release of radioactive waste from land-based sources into the northern seas. National legal regulation of such sources is and will remain very important.

This discussion led the policy working group to make the following recommendation:

- o Arctic states should speed the clarification and further development of principles of responsibility and liability in a framework of international law for environmental damage through effective, appropriate, international bodies.
- o Consideration should be given to the preparation of specific regional/international convention on the protection of the Arctic environment, possibly through the offices of the proposed Arctic Council.

**Financial Issues:**

A number of participants made clear that Russia does not itself possess the financial resources to clean up its existing dump sites in the Arctic. They argued that cleanup is required and that it is in the environmental security interests of other neighbouring countries to assist Russia financially in the process of assessing and managing the problem.

On another financial matter, noting the pressing need for additional data and scientific synthesis to permit policy-makers to arrive at well informed decisions, the working group concerned with policy made the following recommendation:

- o National governments, international bodies such as the World Bank and Global Environment Facility, and private funding agencies, should make available adequate financial resources to support the scientific, technical and social research required for responsible decision-making on the problem of radioactive waste in the Arctic.



**Reducing Future Inventories of Radioactive Waste**

Adopting a preventative perspective, the policy working group made the following recommendation:

- o In the context of the ending of the Cold War, national governments should seek effective ways to limit and reduce military and civil activities which produce radioactive waste in the Arctic Ocean.

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## **Session 1: Inventory**

**Texts Accompanying Presentations**

## **Novaya Zemlya Test Site and the Problem of the Radioactive Pollution of the Polar Ocean.**

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At the present time there are several global sources of the radioactive pollution of Arctic seas and the Polar ocean:

- sunk nuclear reactors and containers with the radioactive waists, sunk nuclear submarines with the nuclear devices and discharge of the liquid radioactive waists into the Arctic seas;
- the nuclear ships (submarines, nuclear icebreakers), navy bases and enterprises of the nuclear-industrial complex functioning in the polar zone;
- radioactive substances discharge in the Arctic seas by the streams of the Siberian rivers (Enisei, Ob with the Irtysh tributary etc.);
- nuclear weapons tests at the archipelago Novaya Zemlya (NZTS) and more than 20 peaceful nuclear explosions in the northern part of Russia (behind the Polar Ring).

The objective of this work is to determine the contribution of the NZTS nuclear weapons tests into the global pollution of the North Polar Basin, to tell about real levels of the radioactive pollution of the environment of the region where those tests were held. The influence of the NZTS nuclear tests on the global pollution of the North Polar basin region is investigated poorly for the present time because of the inaccessibility of the data. But now there is a possibility to provide such investigation because some of the materials on the NZTS tests were published recently [1, 2]. According to the [2] common volume of the nuclear tests at NZTS can be described by the following data:

- 3 underwater nuclear explosions (21.09.55; 10.10.57; 23.10.61);
- 87 nuclear explosions in the atmosphere during 1957-1962, among them there were 83 explosions above the ground and 3 explosions above the water, 1 nuclear device was exploded on the surface of the ground and as the result of this explosion crater with the diameter of 80 m and the depth of 15 m appeared;
- 42 underground nuclear explosions (UNE) were held during the period 1964-1990; there distribution per year is at the fig.1

Thus 132 tests were held during 1955-1990.

The most important episodes in the chronology of the nuclear tests in the atmosphere at NZTS during 1957-1962:

- a. First nuclear explosion at NZTS was held under the water (21.09.55) at the depth of 50 m at Gouba Tchernaya.
- b. Intensive period of the tests:

- 1958: 26 nuclear tests in the atmosphere;
- 1961: 23 nuclear tests in the atmosphere and 1 nuclear test under the water;
- 1962: 36 nuclear tests in the atmosphere.

Total yield equivalent on the division reactor for this period; ~90 Mt.

c. 30.10.61 explosion of the hydrogen superpowerful 50 Mt bomb at the height of 3.5 km.

Total energy of the nuclear tests on the archipelago of Novaya Zemlya was:

- total yield of air and underwater explosions.....239 Mt,
- total yield of underground nuclear explosions.....25 Mt
- total yield .....264 Mt

This is the comparison of these data with the analogous data from the Semipalatinsk test site (STS) and peaceful nuclear explosions (PNE).

Test Sites	NZTS	STS	PNE
Number of tests	132	467	115
total yield (Mt)	264	16	1.5
deposit	94%	4.5%	1.5%

It is evident that 94% of the total yield of FSU nuclear tests is for the NZTS tests. Total yield of air and underwater nuclear tests at NZTS approximately twice more than the total yield of USA, England, France nuclear tests in the atmosphere, space and water (272 tests during 1945-1962, ~100Mt) and China (22 tests during 1964-1980, 12.7 Mt). According to these data it is evident that main pollution of the environment with radioactive waists of nuclear explosions is caused by the nuclear weapons tests at NZTS and especially by atmosphere nuclear tests.

The square of the NZTS is  $55 \cdot 10^3 \text{ km}^2$ , the square of the Novaya Zemlya Archipelago is  $82 \cdot 10^3 \text{ km}^2$ . Nuclear weapons tests at the NZTS were held at the three flats (fig.2):

- at the South part of the Novaya Zemlya Archipelago in the region of the Couba Tchernaya during 1955-1962 surface explosion (07.09.57) and atmosphere explosions above the water and under the water were held, during 1972-19975 underground tests were held in the holes (flat A);
- at the north island in the region of the Soukhoi Nos peninsula and Gouba Mitioushikha during 1957-1962 atmosphere tests were held and in particular explosion with the yield 50 Mt (flat C);
- in the north part of the south island in the region of the Matochkin Shar strait during 1964-1990 nuclear underground tests were held in horizontal holes (flat B).

According to consequences of radioactive pollution there are five regions at the A flat:

1. Radioactive trace of the underwater nuclear explosion held in 1955 with the width of 2 km approximately and the square of several square kilometers crosses the Koushny peninsula in the south direction. Pollution caused by radionuclides Cs-137, Sr-90 and Co-60 that are spread

in the upper layer of the soil that is 6-10 cm thick. At the present time maximum levels of the activity are 30  $\mu\text{R/h}$ , variation of the pollution density is 0.8 - 13  $\text{ci/km}^2$

2. Radioactive trace of the surface explosion held in 1957 goes from the south shore of the Gouba Tchernaya to the east, crosses the whole south island, Yamal and Taymuir peninsulas. Right after the explosion radioactive cloud trace within the distance of approximately 1500 km to the Earth. Pollution of the district with the radius of 400 m around the crater is characterized now by the activity levels 1000 - 15000  $\mu\text{R/h}$  and caused by the radionuclides Sr-90, Cs-137, Co-60, Eu-152, Pu-239. Density of the Cs-137 pollution is 0.25 - 30  $\text{ci/km}^2$  and Sr-90 is 0.04 - 15  $\text{ci/km}^2$ . At the 30 km distance density of these radionuclides pollution is approximately 0.07  $\text{ci/km}^2$ .
3. Radioactive trace of the low air explosion at the present time is an area with the diameter of approximately 0.5 km with the activity level approximately 30  $\mu\text{R/h}$  and density of the pollution by Eu-152 0.05 - 3.5  $\text{ci/km}^2$ , Co-60 approximately 0.6  $\text{ci/km}^2$ , Sr-90 and Cs-137 approximately 0.05  $\text{ci/km}^2$  each.
4. Radioactive trace of the explosion held above the water goes from the Gouba Tchernaya to the north-east and is characterized at the present time by the activity levels 20-25  $\mu\text{R/h}$  and density of the pollution by Sr-90 and Cs-137 0.1 - 1.2  $\text{ci/km}^2$ .
5. Radioactive trace of the underground explosion in the hole U-4 in 1973 is created by the escape of gas products in 20 minutes after the explosion and goes into the south-east direction. At the present time levels of the activity at this trace near the hole are 25  $\mu\text{R/h}$  and total activity of Cs-137 created after the disintegration of xenon-137 not more than 1 Curie.

At the C flat 4 regions of radioactive pollution consequences were determined by the radiation filming and radiochemical analysis of the soil. According to the results of the measurement of the density pollution held in 1992 radionuclide composition, activity levels and sizes of the polluted districts are in the table.

No	Region	Area ( $\text{km}^2$ )	Activity level ( $\mu\text{R/h}$ )	Cs-137 ( $\text{Ci/km}^2$ )	Eu-152 ( $\text{Ci/km}^2$ )	Co-60 ( $\text{Ci/km}^2$ )
1	East part	0.4	30-40	0.07	0.15-0.6	0.05
2	Center part	0.3	25-35	0.06	0.45	-
3	West part	0.5	25-30	0.5	-	-
4	North part	0.3	20-25	0.05	-	-

At the B flat only UNE were held in horizontal holes during 1964-1990. During UNE radioactive products of the explosion are concentrated basically inside the massif in the cavity, melting lens, collapse pillar and destruction zone. Relative spreading of the radioactive products of the explosion in the environment are approximately as following:

- cavity and melting lens ..... 70-90%
- collapse pillar and destruction zone..... 10-30%
- radioactive gazes escape into the atmosphere..... 1-10%

Level of the radioactive pollution of the atmosphere and the surface of the Earth during the UNE

is determined by the amount of radioactive products of the explosion in the escaping gases. Isotope composition and quantity of the activity escaped depend on the time between the moment of the explosion and appearance of radioactive products in the atmosphere and also on the yield of the explosion. All UNE held at the NZTS according to their radioactive consequences and levels of the rest radioactive pollution can be divided into following groups:

- 15 tests (36%) - radioactive products almost completely located inside the rock massif (debris, gases containment);
- 25 tests (60%) - were accompanied by filtration escape of radioactive gases in the atmosphere (seepage of radioactive inert gases);
- 2 tests (4%) - were accompanied by the dynamic pressure escape in the atmosphere of gas and steam products (dynamic venting of radioactive gases into the atmosphere). In this last case amount of the escaping activity reached its maximum level and there was quite dangerous situation for the participants of the test when radioactive cloud with high levels of the activity and long radioactive trace appeared. According to the calculations total radioactive pollution of the NZTS territory during all UNE by the disintegration of xenon-137 penetrated into the atmosphere can be characterized by the drop of 2500-5000 Ci of Cs-137.

Territory of the B flat is under the constant radiation monitoring of the experts from the different organizations. Here are the results of the measurements of the radiation situation at the B flat in terms of the pollution density of the surface of the Earth.

No	Region	Cs-137 (Ci/km <sup>2</sup> )	Sr-90 (Ci/km <sup>2</sup> )
1	North part of Mt.Moiseev	0.044	0.052
2	West part of Mt.Chernaya	0.1	0.054
3	North part of Mt.Lazareva	0.05	0.076
4	Epicentral part of Mt.Lazareva	0.67	0.079
5	Valley Shoumilikha	0.08	0.052

Maximum levels of the activity in several places close to the epicenters of the explosions reach 100  $\mu$ R/h, for the rest of the flat B territory background meanings of the activity are 10-20  $\mu$ R/h. According to the following data it is evident that most densities of the pollution and activity levels are at the A flat - they are 10 to 10<sup>2</sup> higher than at the C and B flats. The widest composition of the anthropogenic radionuclides including Pu-239 was determined at the A flat.

These materials on the radioactive pollution and the others received during the tests allow in future to held direct calculations of the NZTS nuclear tests contribution in the global pollution taking into consideration all possible mechanisms of the radionuclides discharge into the Ocean: spreading by the atmosphere streams, drops from the radioactive cloud and later discharge by the surface waters, migration with the underground waters. Contribution of each mechanism is important at different stages of the development of the global pollution process and must be considered as a separate objective of the investigation. We have to notice that participation of the underground waters in the spreading of the radionuclides at the NZTS is quite difficult because of the deep layer (300-500 m) of the eternal frost and small thickness of its melting during 3-4 summer

months. As far as the discharge of radionuclides by spring waters and river streams directly from the tests flats of NZTS is concerned (r. Shoumilikha and so on), regular observations prove that at the present time discharge of radionuclides into the Barents sea is decreased a lot, but Tritium discharge is approximately  $10^3$  Ci/year, that is  $10^3 - 10^4$  higher than Sr-90 and Cs-137 discharge.

In conclusion we would like to suggest rough estimation of the integral contribution of two first mechanisms of radionuclide discharge into the global radioactive pollution of the North Polar region. That is why chronological comparison of the nuclear explosions energetic pressures with the published data by the global drop of Sr-90 [3] was held. At the fig.3 annual drops of Sr-90 at the territory of the North Hemisphere during 1961-1981 is pointed out and there are also total yields of the nuclear explosions in the atmosphere and water (period I), of the Mt class UNE up to 1975 (period II) and of the UNE with yields limited by the threshold of 150 Kt (period III). The direct connection between the global Sr-90 drop and nuclear tests in the atmosphere during 1957-1962 is evident. After the cessation of the tests in the atmosphere, space and water in 1963 considerable decrease of radioactive drops from the atmosphere is observed. When UNE with the high yields were started the decrease of the annual Sr-90 drops stopped at the approximate level  $2 \cdot 10^5$  Ci/year up to 1970. During next years the connection between global Sr-90 drops and nuclear tests at the NZTS is not so evident and has to be analyzed in details. The same comparison of the published data on the global Sr-90 discharge into the Polar ocean [4,5] with the periods of intensive nuclear tests at the NZTS is pointed out at the fig.4. In this case correlation between NZTS tests and radioactive Sr-90 discharge and correlation between Sr-90 discharge into the Polar ocean and global drops from the atmosphere can be noticed only during the period of the intensive nuclear tests in the atmosphere.

### Conclusions:

1. Global pollution with the radionuclides from the NZTS of the North Polar region took place mostly during the period of intensive nuclear tests in the atmosphere and in the water (1955-1965). During this period global Sr-90 drop and its discharge into the Polar ocean increased in  $10-10^2$  times.
2. At the present time medium quantity of the surface density pollution at the NZTS by Cs-137 is  $0.09 \text{ Ci/km}^2$  ( $0.33 \text{ Bq/cm}^2$ ) and Sr-90 is  $0.07 \text{ Ci/km}^2$  ( $0.22 \text{ Bq/cm}^2$ ).
3. At the UNE during 1964-1990 main mass of the radioactive products was located under the ground in the source of the explosion and only small part of the activity (1-10%) that led to the radioactive pollution mostly of the NZTS territory escaped into the atmosphere.

### Literature:

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### **Figure Captions:**

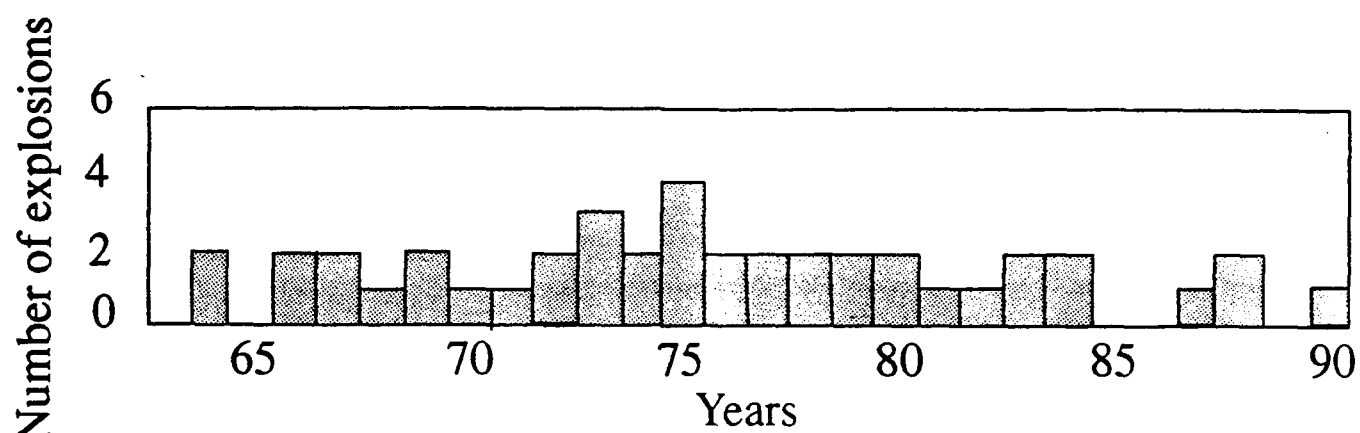
Fig.1. Distribution of the number of UNE's per year.

Fig.2. Different tests flats at the NZTS.

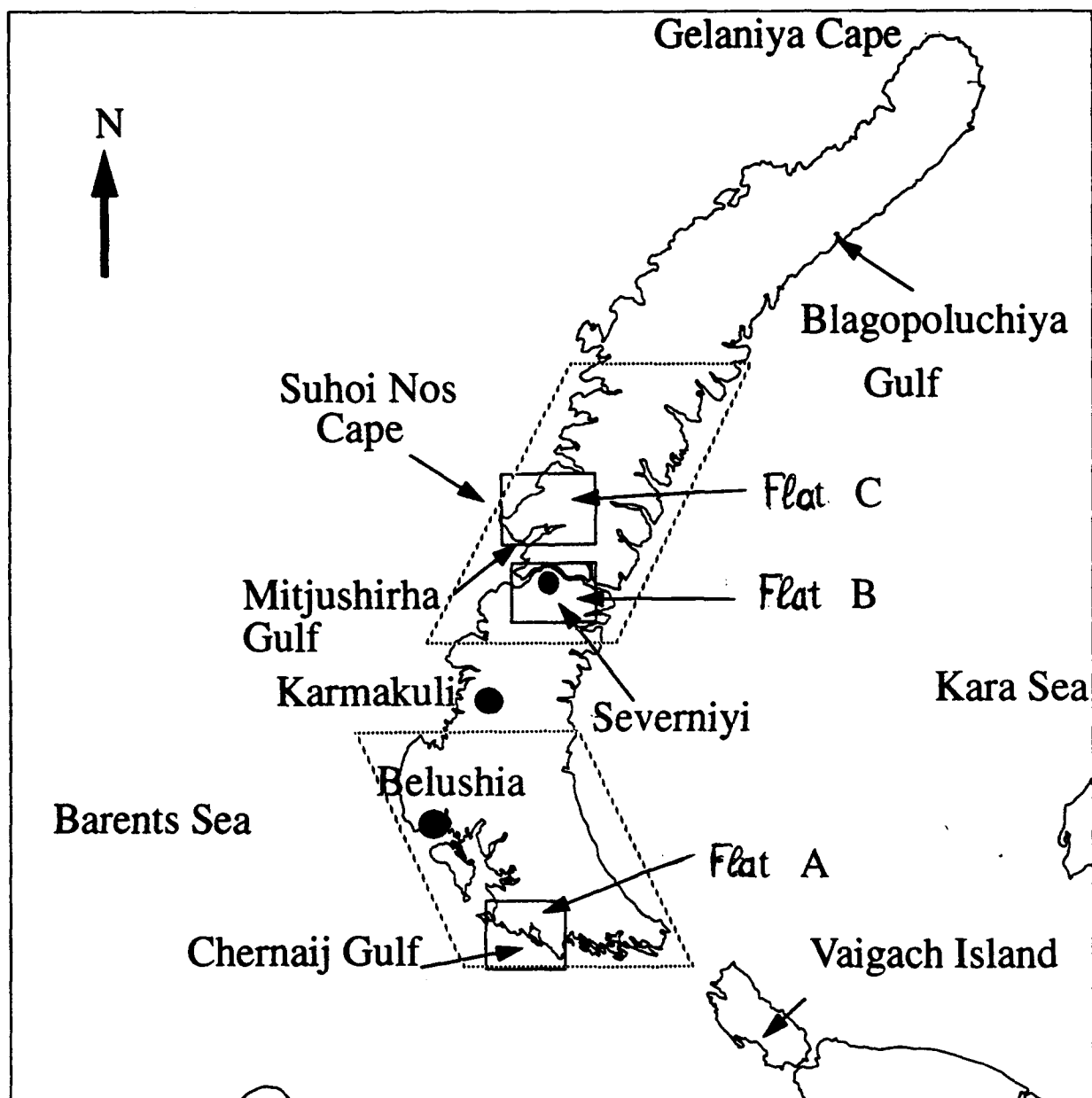
Fig.3. Comparison of the annual Sr-90 drop at the territories of the North Hemisphere with the main periods of the tests and total yield of the explosions.

Fig.4. Comparison of the annual Sr-90 discharge into the North Polar Basin with the main periods of the tests and total yield of the explosions.



*Fig. 1*

## DIFFERENT TESTS FLATS AT NZTS

surface of NZTS:  $55.10^3 \text{ km}^2$ surface of the archipelago:  $82.10^3 \text{ km}^2$ 

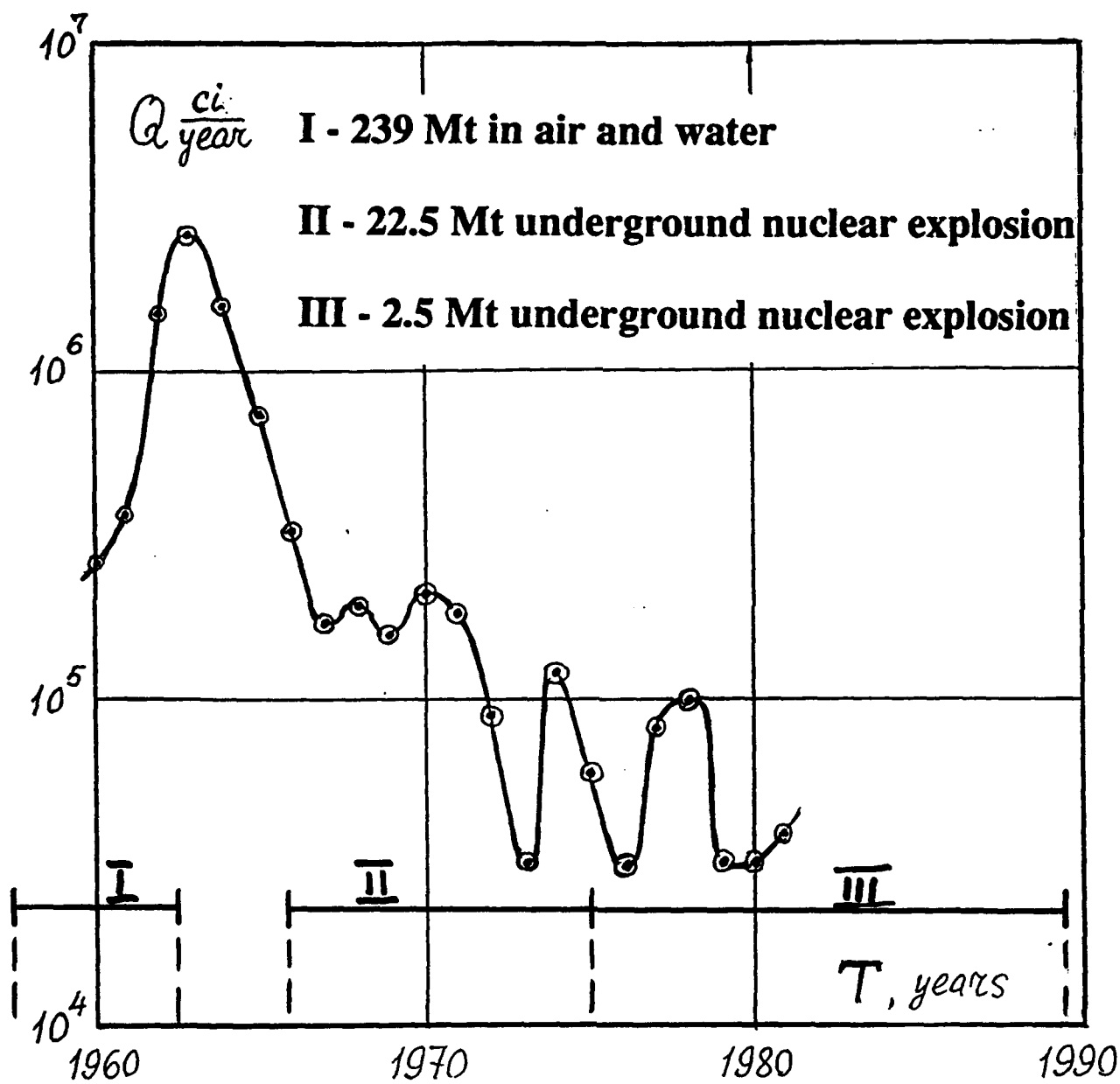


Fig. 5

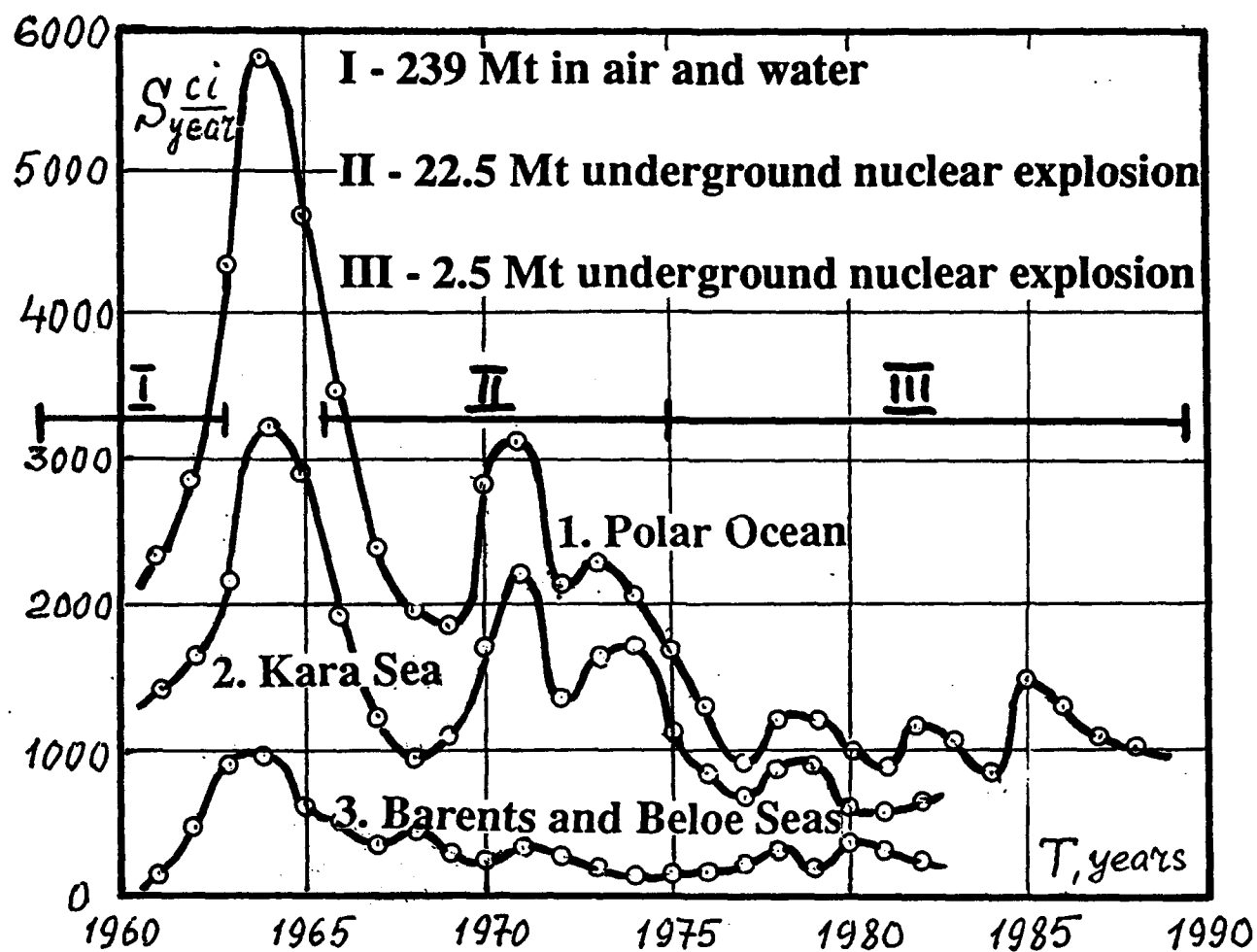


Fig 4

## **Sediment Inventory of $^{137}\text{Cs}$ in the Eastern Chukchi sea, Northwest Alaskan Arctic**

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### **ABSTRACT**

Cesium-137 concentrations were measured in six sediment core samples collected from the Chukchi sea and Kotzebue Sound areas of the Alaskan Arctic. The specific concentration of  $^{137}\text{Cs}$  varies between below detection limit to 333 dpm/kg on a dry weight basis when corrected to January 1, 1990. The sediment inventory values range between 0.64 and 4.09 dpm/cm<sup>2</sup> with a mean of 2.18 dpm/cm<sup>2</sup>. The penetration depths of  $^{137}\text{Cs}$  in the sediments were always greater than that of excess  $^{210}\text{Pb}$ . The mean inventory of  $^{137}\text{Cs}$  is comparable to the expected inventory of 2.70 dpm/cm<sup>2</sup>, suggesting that there is very little, or no input to the sampling sites other than Pre-Chernobyl radioactive fallout.

### **INTRODUCTION**

Movement of particle-associated radionuclides in the Arctic, where sediment transport by sea-ice may be involved, has not been well studied. Both the lakes and seas in the Alaskan Arctic are ice covered for 7-8 months per year, which precludes the atmospherically-delivered radionuclides from directly reaching the water-column most of the year. Removal of ice-cover leads to the redistribution of ice-rafted sediments as well as the associated sediment-sorbed particle-reactive radionuclides (Nichols, 1967; Hermanson, 1990) and their eventual deposition.

Earlier studies on the Arctic Basin sediments have shown that the inventories of particle-reactive nuclides such as  $^{230}\text{Th}$ ,  $^{231}\text{Pa}$  and  $^{10}\text{Be}$  are much lower than what is to be expected from either the production of these nuclides from their parents ( $^{230}\text{Th}$ ,  $^{231}\text{Pa}$ , Ku and Broecker, 1966; Moore and Smith, 1986; Bacon et al., 1989) or from the cosmogenic production rate ( $^{10}\text{Be}$ , Finkel et al., 1977). These low inventories were attributed to poor scavenging efficiencies of the glacio-marine sediments (due to their

low ion-exchange capacity; Naidu et al., 1984), and low particle concentration in the water column (Bacon et al., 1989). Data on the sediment inventories of particle-reactive nuclides from the Arctic continental shelves are either scarce or do not exist. Further, very limited data exist on the distributions of anthropogenic radionuclides ( $^{137}\text{Cs}$ ,  $^{239,240}\text{Pu}$ ,  $^{241}\text{Am}$  etc) in marine sedimentary column of the northern high latitudes ( $\geq 66^\circ \text{N}$ ). These nuclides were introduced into the Arctic waters by bomb fallout and discharge of nuclear-wastes from areas of Former Soviet Union as well as from the British Nuclear Fuels Ltd at Sellafield.

There is a fairly large data base on the dispersion of radionuclides discharged into the Irish Sea from the British Nuclear Fuels Ltd at Sellafield (Livingston and Bowen, 1977; Mackenzie and Scott, 1982 and the references therein). Recent revelations show that a significant portion of former Soviet Union's Arctic coastal waters and sediments are polluted by nuclear reactor-derived radionuclides, which is of environmental concern. Initial contaminant transport estimates from oceanographic circulation models suggest that these radionuclide contaminants could be transported across the Russian Arctic Ocean in a time scale of 3-5 years. Thus, these radionuclides could, in principle, reach the Alaskan shelves from the western Bering or East Siberian seas. However, the extent of this contamination and its impact on the biological community is unknown. The objective of this paper is: (i) to investigate the  $^{137}\text{Cs}$  sediment inventory in the eastern Chukchi sea, Alaskan Arctic and compare this inventory to the value estimated from the atmospheric bomb fallout for this region and (ii) to assess if there is any contribution of  $^{137}\text{Cs}$  following the former Soviet Union's nuclear waste dump sites and/or atmospheric fallout subsequent to the Chernobyl accident. In addressing these, we report the concentrations and inventories of  $^{137}\text{Cs}$  in six sediment cores from the Northwest Alaskan shelf.

## **STUDY AREA**

The study area consists of the East Chukchi Sea, Alaskan Arctic (Fig. 1). The environmental attributes of the study area have been described in Feder et al. (1990). Briefly, the study area consists of a broad, relatively shallow (~50 m) and flat shelf with minor relief generated by ice gouging. The typical climate is the presence of long, severely cold winters with ice cover for about 7-8 months and short, cool summers for the rest of the year. The sea floor consists of a mosaic of sediment types, with a broad across-the shelf gradation of sandy gravels in shore to sandy muds at the central shelf (Naidu, 1988). Major component of the sediments are river-discharged. Coastal erosion

in the Alaskan Arctic is unusually high ( $\sim 1$  m/yr; Naidu et al., 1984). Sea ice plays a dominant role in sediment transport.

## **MATERIALS AND METHODS**

Six sediment core samples were collected in 1986 and 1987 from the East Chukchi Sea (Figure 1, Table 1) on board R/V *Oceanographer* and *Surveyor* using a Benthos gravity corer (6.7 cm i.d. plastic core liners). Additional box core samples (84-12) were provided by Dr. L. Phillips of the U. S. Geological Survey. All cores were stored frozen until ready for analysis. Each core sample was extruded out of the plastic liner and split into 1-cm sections and the water contents were determined after drying at  $90^\circ$  C for about 24 hours. The porosities were calculated from the water content, assuming a dry sediment density of  $2.5 \text{ g cm}^{-3}$ .

Most of the analyses were begun several months after collection. The details on the  $^{210}\text{Pb}$  profiles are given elsewhere (Baskaran and Naidu, 1993, in preparation).  $^{137}\text{Cs}$  activity concentrations were determined from gamma-counting of 10-30 g of dried sediment with a Ge(Li) detector coupled to a multichannel analyzer. The gamma detector counting efficiency at 661.6 keV (Gamma ray yield = 85.1%) was determined with a NBS river sediment standard for various geometries. The background and counting efficiencies for various geometries, are referred to in Baskaran et al. (1988).

## **RESULTS AND DISCUSSION**

The surficial  $^{137}\text{Cs}$  specific concentration varies between 146 and 333 dpm/kg with a mean of 191 dpm/kg. These  $^{137}\text{Cs}$  concentrations were all decay-corrected to 1st January 1990. The specific concentration is plotted against depth for all 6 sediment cores in Figure 2. The  $^{137}\text{Cs}$  concentration maximum does not correspond to any particular depth in all the six sediment cores and it varies from core to core (Fig. 2). The maximum  $^{137}\text{Cs}$  concentration occurs between 1 and 2 cm in four of the 6 sediment cores. In the remaining two cores, the  $^{137}\text{Cs}$  maximum occurs between 5 and 7 cm (Fig. 2).

### **Total Sediment Inventory ( $I_T$ ) of $^{137}\text{Cs}$ in the Chukchi and Kotzebue Sound regions:**

The  $^{137}\text{Cs}$  inventories in the six sediment cores are given in Table 1. The  $^{137}\text{Cs}$  inventory varied between 0.64 and  $4.09 \text{ dpm/cm}^2$  with a mean of  $2.18 \text{ dpm/cm}^2$ . The  $^{137}\text{Cs}$  penetration depths in the cores, which varied between 5.5 and 16.5 cm, are plotted against excess  $^{210}\text{Pb}$  ( $^{210}\text{Pb}_{\text{xs}}$ ) penetration depths (Fig. 3). In four out of five sediment

Table 1: Sediment core locations, water column depth, mud content, penetration depths of  $^{210}\text{Pb}$  and  $^{137}\text{Cs}$  sedimentary inventory of  $^{137}\text{Cs}$

Sediment core*	Coordinates	Mud content (%)**	Penetration depth of $^{210}\text{Pb}_{\text{xs}}$ (cm)	Penetration depth of $^{137}\text{Cs}$ (cm)	$^{137}\text{Cs}$ inventory (dpm/cm <sup>2</sup> )
CH-39 (48)	71°52.2' N 168° 15.4' W	95.7 (32.5)	9.0	10.5	$0.64 \pm 0.04$
CH-40 (45)	70° 16.7' N 167° 54.3' W	47.2 (19.2)	6.5	5.5	$0.92 \pm 0.07$
SU-5 (50)	67° 02.3' N 169° 00' W	43.0 (9.6)	8.5	13.5	$2.48 \pm 0.16$
KS-1 (13)	66° 37.5' N 162° 59' W	42.9 (17.3)	7.5	8.5	$2.49 \pm 0.16$
SU-87-10 (42)	67° 04.4' N 166° 47.5' W	NM	>20	16.5	$2.44 \pm 0.08$
84-12 (42)	70° 9.33' N 166° 34' W	NM	11.5	15.0	$4.09 \pm 0.20$

NM: Not measured

\* Numbers in parenthesis denote water depth (in meters)

\*\* Numbers in parenthesis denote percentage of clays ( $\leq 4 \mu\text{m}$ )

cores for which the penetration depths of both the nuclides are available, the  $^{137}\text{Cs}$  penetration depths are greater than that of  $^{210}\text{Pb}_{\text{xs}}$ . This observation is generally attributed to the relatively greater depositional downward mobility of  $^{137}\text{Cs}$  than  $^{210}\text{Pb}$  which is consistent with results reported elsewhere (Edgington, 1981; Beasley et al., 1982; Santschi et al., 1983).

The diffusion coefficient for Cs ion is related to  $\phi^2$  (where  $\phi$  is the porosity), (Lerman, 1979) for metal ions. Also, the partition coefficients between seawater and sediments are 100-500 (Nyffeler et al., 1984; Li et al., 1984) and thus pore water movement could easily enable the Cs ions to move deeper than the Pb ions.

#### Contribution of Chernobyl-derived radiocesium to the Alaskan Arctic continental shelf sediments :

The sources for the sedimentary  $^{137}\text{Cs}$  at the study-site could have been derived from three different sources: (i) Pre-Chernobyl fallout ( $I_{\text{PC}}$ ) (ii) Chernobyl accident ( $I_{\text{C}}$ ) and (iii) Reactor dump sites in the Former Soviet Union's coastal waters ( $I_{\text{RFS}}$ ). Quantitative information on the importance of these sources needs to be evaluated. The



total sediment inventory ( $I_T$ ) =  $I_{PC} + I_C + I_{RFS}$ . The reactor-derived  $^{137}\text{Cs}$  contribution can be estimated, if the total sediment concentrations, amounts of Pre-Chernobyl and Chernobyl-derived concentrations are known.

It is estimated that the Chernobyl accident on 26 April 1986 was at most 5-10% of the total amount of radiocesium released from all atmospheric nuclear weapons tests (Goldman, 1987). Highly variable regional deposition rates of Chernobyl-derived radionuclides have been reported from Scandinavia, Germany, the United Kingdom and the Mediterranean region, as suggested by the wide-ranging concentrations of  $^{137}\text{Cs}$ , for example, from 1 to several thousand  $\text{nCi/m}^2$  (Davidson et al., 1987). The total deposition rate of  $^{134}\text{Cs}$  ranged between 1-2  $\text{nCi/m}^2$  in the western United States while at several

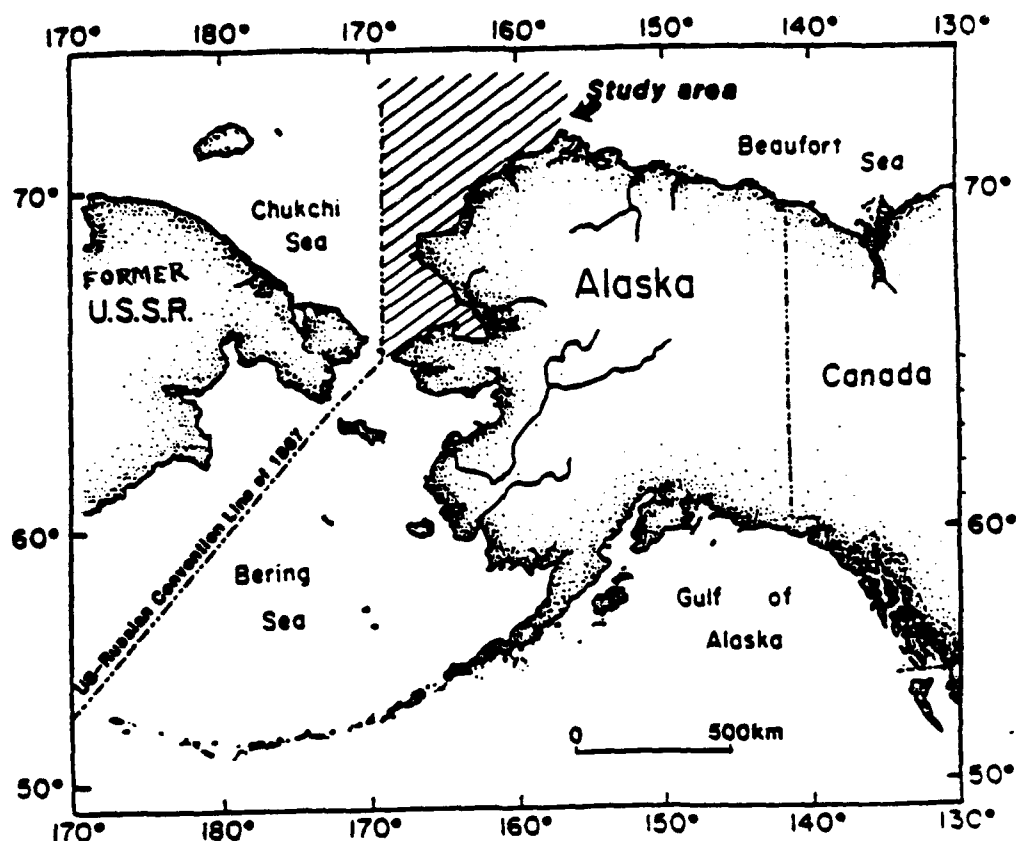


Figure 1: The study area in the northeastern Chukchi Sea and Kotzebue Sound area as shown by the shading on the map.

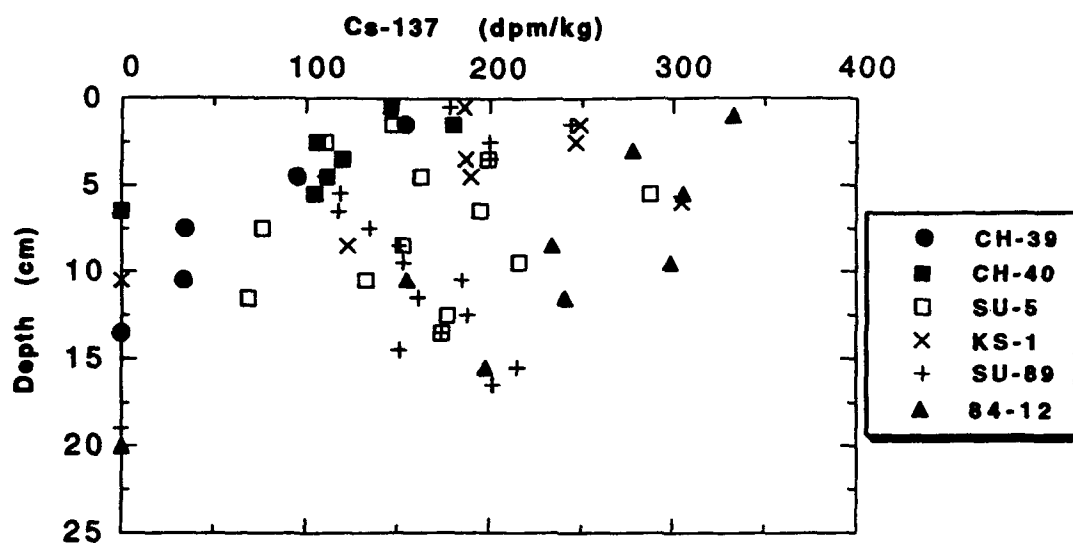


Figure 2:  $^{137}\text{Cs}$  concentration profiles in eastern Chukchi sea sediments, Northwest Alaskan Arctic.

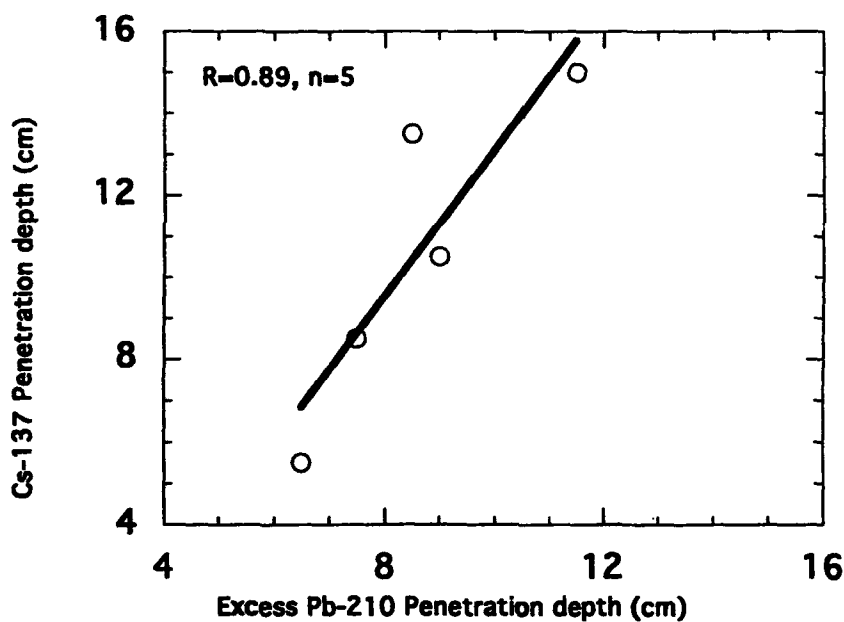


Figure 3:  $^{137}\text{Cs}$  penetration depth is plotted against  $^{210}\text{Pb}_{\text{ex}}$  penetration depth. This plot shows that there is post mobility of  $^{137}\text{Cs}$  after its deposition.

locations in the Midwest and on the east coast, the values ranged from 0.1 - 1 nCi/m<sup>2</sup> (Davidson et al., 1987). The radiocesium deposition rate in Alaska was about two orders of magnitude lower than that of the western United States, estimated at approximately 0.014 nCi/m<sup>2</sup> for April 1986 (White et al., 1986; Baskaran et al., 1991). Based on <sup>134</sup>Cs measurements on soil samples, it is indicated that there was a wide regional difference in the deposition rate of the Chernobyl-derived radionuclides in Alaska (Baskaran et al., 1991). The regional difference could be linked to the wide differences in the Chernobyl-derived radionuclides over Alaska, as reflected by the regional concentrations of integrated radioiodine (<sup>131</sup>I) concentrations in the air samples in Fairbanks and Barrow (Kipphut, 1986 - unpublished data). This difference in the radionuclide concentration in air samples could lead to differences in the deposition rates.

Very limited data exist on the Chernobyl-derived fallout of radiocesium in the coastal Alaskan waters. The Chernobyl-derived radiocesium concentrations in a variety of marine biological tissues and lake deposits are very low. The Chernobyl-derived radiocesium concentrations on two fish samples and two Bowhead whale samples collected from Arctic waters (fish from Colville River 70.0° N, 151.0° W and Bowhead Whale from Arctic Ocean, 71.0° N, 157.0° W) are available. The <sup>137</sup>Cs concentration was below detection limit in Bowhead whale blubber and liver, while the muscle sample had a low concentration of <sup>137</sup>Cs (15.4 pCi/kg). Since the fallout is greatly diluted in the ocean water and the Bowhead whale's prey is unlikely to concentrate <sup>137</sup>Cs, higher concentrations were not found in the marine mammals. When the Chernobyl plume arrived the Northern Alaska and adjoining oceanic waters, snow cover existed on the Arctic soil and water. Most of these snow-deposited radiocesium would have been washed away when the snow melted and we would expect to find high concentrations of radiocesium in lakes in that area. In one grab sediment sample collected from a small lake (71.3° N, 156.2° W, collected in 1988), the <sup>134</sup>Cs level was below detection limit (Baskaran et al., 1991).

We also noted that suspended marine sediments collected in sediment traps in 1987 from the Eastern Chukchi sea also did not contain measurable <sup>137</sup>Cs or <sup>134</sup>Cs concentrations suggesting that the suspended particles in those areas also did not sequester measurable Chernobyl-derived radiocesium. Thus, it is inferred that the sedimentary inventory of radiocesium in the Chukchi is primarily pre-Chernobyl derived.

#### **Expected Inventory of <sup>137</sup>Cs from pre-Chernobyl period in the study area:**

<sup>137</sup>Cs fallout data from the study area is not available. However, one can estimate its likely value indirectly, from a proxy of the latitudinal distribution of <sup>90</sup>Sr fallout

between 1958 and 1967 as well as from the relationship between  $^{90}\text{Sr}$  fallout and latitudinal distance north from International Falls, Minnesota. Sugai (1990) estimated the expected inventory of  $^{137}\text{Cs}$  to be  $15.3 \text{ dpm/cm}^2$  for  $55^\circ \text{ N}$  for 1982. From the published data on the atmospheric deposition of  $^{90}\text{Sr}$  at different latitudes for the years 1958-1967 (Joseph et al., 1971), the expected fallout of  $^{90}\text{Sr}$  (or by implication  $^{137}\text{Cs}$ ) at  $70^\circ \text{ N}$  (our study area) is estimated to be about 21.5% as that of  $55^\circ \text{ N}$ . Thus, the expected sedimentary inventory of  $^{137}\text{Cs}$  in 1990 at  $70^\circ \text{ N}$  would be  $2.74 \text{ dpm/cm}^2$ . However, most of the fallout radiocesium do not readily reach sediment-water interface due to its longer residence time in the oceanic water-column and a major portion of it tends to remain in the water column. The river-discharged sediments will also contain relatively higher concentrations of radiocesium and thus the input to the sediment is not strictly from the overhead fallout. Using the relationship between total  $^{90}\text{Sr}$  fallout and distance north from International Falls (see HASL, 1977), one can determine the  $^{90}\text{Sr}$  fallout at any distance from the International Falls. The study site is about 3100 km away from the International Falls. This distance corresponds to  $11 \text{ mCi/km}^2$  ( $1 \text{ mCi/km}^2 = 0.222 \text{ dpm/cm}^2$ ) or  $2.44 \text{ dpm/cm}^2$  of  $^{90}\text{Sr}$  in 1976. Assuming a value of 1.5 to be the ratio of  $^{90}\text{Sr}/^{137}\text{Cs}$  (HASL, 1977; Hermanson, 1990), the sedimentary inventory of  $^{137}\text{Cs}$  in 1990 at  $70^\circ \text{ N}$  is expected to be  $2.65 \text{ dpm/cm}^2$ , which is in close agreement with the value estimated previously. In other words, the expected Pre-Chernobyl sediment inventory can be taken to be  $2.70 \text{ dpm/cm}^2$ .

#### **Radiocesium derived from reactor-derived Soviet coastal waters (IRFS):**

As stated earlier, in the Chukchi sea, the mean inventory of  $^{137}\text{Cs}$ ,  $2.18 \text{ dpm/cm}^2$  (range: 0.64 and  $4.09 \text{ dpm/cm}^2$ ), can be compared to the Pre-Chernobyl inventory of  $2.70 \text{ dpm/cm}^2$ . It has been estimated from simple physical hydrographic circulation models that the reactor-derived contaminant radionuclides from the Russian Arctic waters could even be transported across the Russian Arctic Ocean in a short time scale, 3-5 years. Since a significant portion of the fallout radiocesium at the study sites would not directly reach the sediment and there is an additional input from the river-discharged sediments, it is difficult to quantitatively evaluate the amount of radiocesium that might have been derived from the nuclear-reactor dump sites. Since the present  $^{137}\text{Cs}$  inventory is comparable to that of pre-1986, it is likely that very little or none of the Chernobyl and/or more recent reactor-derived radionuclides has deposited within the Chukchi sea sediments. This could be attributed to the discharge of negligible amounts of the radionuclides into the eastern Chukchi Sea.

## **CONCLUSION**

The following conclusions are drawn from data on the sediment inventory of  $^{137}\text{Cs}$  for the northwestern Alaskan Arctic continental shelf sediments:

- (i) The  $^{137}\text{Cs}$  inventory in the year 1990 varied between 0.64 and 4.09 dpm/cm<sup>2</sup> with a mean of 2.18 dpm/cm<sup>2</sup>.
- (ii) The  $^{137}\text{Cs}$  penetration depth varied between 5.5 and 16.5 cm. Generally, the  $^{137}\text{Cs}$  penetration depth is greater than that of  $^{210}\text{Pb}_{\text{xs}}$ . This is consistent with the generally higher post depositional mobility of Cs.
- (iii) The mean measured  $^{137}\text{Cs}$  inventory (2.18 dpm/cm<sup>2</sup>) is comparable to the expected inventory (2.70 dpm/cm<sup>2</sup>) obtained based on the total fallout of  $^{137}\text{Cs}$  in the study area. Close agreement between these two values suggests that either there is apparently very little or no input into the eastern Chukchi Sea from the former Soviet Union's nuclear waste dump-sites in Kara and/or Barents Seas.

## **ACKNOWLEDGMENTS**

We thank Larry Philip of USGS for providing a box core sample from the study area. This study was funded by the Minerals Management Service, Department of the Interior, through an interagency agreement with the National Oceanic and Atmospheric Administration, Department of Commerce, as part of the Alaska Outer Continental Shelf Assessment Program, Drs. H. M. Feder and A. S. Naidu of the University of Alaska Fairbanks through a co-operative agreement (NA-ABH-00031). Part of the sample analyses was done at Texas A & M University at Galveston and supported by Texas Advanced Research Grant (TAR-003581). We thank George Kipphut for providing his unpublished data on the radioiodine measurements on air filter samples, relating to the Chernobyl incident.

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## ANTHROPOGENIC RADIOACTIVITY IN THE ARCTIC SEAS: TIME TRENDS AND PRESENT LEVELS

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### Abstract

Concentrations in water of  $^{99}\text{Tc}$ , Chernobyl- $^{137}\text{Cs}$  ( $^{134}\text{Cs}$ ), total  $^{137}\text{Cs}$ , and  $^{90}\text{Sr}$  from depth profiles including several water masses at  $71^\circ\text{N}$  between East Greenland and Jan Mayen and in the Denmark Strait from 1988 and 1990 are discussed on the background of changing sources. A major surface water mass in the area is the Polar Water reflecting the radionuclide concentrations in the upper layer of the Arctic Ocean. It appears that the present levels may well be explained by pre-1970 sources including early Soviet discharges and global fallout, the European reprocessing facilities, mainly Sellafield, and a Chernobyl contribution. There is no indication of significant additional sources in the present material. However, the effect of additional sources may be masked under the fluctuating European contribution adding significantly to the uncertainty.

### Introduction

As part of the International Greenland Sea Project (GSP), Joint Icelandic - Danish cruises to the southern Greenland Sea and the Denmark Strait were performed annually 1987 - 1991 by the Icelandic fisheries research vessel "Bjarni Sæmundsson". Cruise reports, oceanographic data and other background material is being published in a series of internal reports (see e.g. Kristmannsson et al 1991, Malmberg & Buch 1992). A monograph on the results from the Danish - Icelandic Greenland Sea Project in which the following will be integrated is in preparation. During the 1988 and 1990 cruises, large volume water samples were analyzed for

Table 1: Characteristics for the 4 measured radionuclides.

Isotope	$T_{1/2}$ , yr.	Radiation	Sources (ranked after importance)
$^{99}\text{Tc}$	210.000	$\beta$	~ 85 % European coastal ~ 15 % weapons tests
$^{137}\text{Cs}$	30	$\gamma, \beta$	Atmospheric weapons tests ~ 35 % European coastal ~ 15 % Chernobyl Russian sources
$^{134}\text{Cs}$	2.1	$\gamma, \beta$	Chernobyl
$^{90}\text{Sr}$	28	$\beta$	Atmospheric weapons tests Russian sources ~ 15 % European coastal

4 man-made radionuclides:  $^{99}\text{Tc}$ ,  $^{137}\text{Cs}$ ,  $^{134}\text{Cs}$ , and  $^{90}\text{Sr}$ . Table 1 summarizes some characteristics for these radionuclides in the study area.

A description of the different water masses in the study area will appear in the above mentioned monograph. The water masses mentioned in the present paper are defined by limits in potential temperature and salinity as set up in Table 2.

Radionuclide results from the 2 cruises are given in Figures 1-3 and in Table 3-4, where mean concentrations of the measured radionuclides in different water masses are given. Here, the measured  $^{134}\text{Cs}$  has been recalculated to "Chernobyl  $^{137}\text{Cs}$ " by applying the observed  $^{134}\text{Cs} / ^{137}\text{Cs}$  ratio of 0.54 by May 1, 1986 (Aarkrog 1988), decaycorrected to the sampling time. The separation of the different water masses are shown in Figures 2 and 3. For clarity, the minor water masses Polar Intermediate Water (PIW) and Arctic Surface Water (ASW) have been included in the Arctic Intermediate Water (AIW) in the present work.

### Sources

The different sources contributing to the present anthropogenic radionuclide concentrations in the Greenland Sea and the Denmark Strait exhibit different characteristics. The global fallout from atmospheric nuclear test explosions contaminated the world ocean during the 1950's and 1960's with a distinct peak 1962-1965. Throughout the 1970's and 1980's the Polar surface water in the East Greenland Current has shown higher concentrations of  $^{137}\text{Cs}$  and  $^{90}\text{Sr}$  than the central North Atlantic in spite of the fact, that high latitude atmospheric global fallout was smaller than mid-latitude fallout. This excess contamination of the Arctic Ocean may partly be explained by direct fallout from the Novaya-Zemlia nuclear test site, by underwater nuclear explosions and dumping of nuclear wastes in the Arctic Ocean shelf seas. During the 1970's and early 1980's, this Polar Water anomaly included excess  $^{90}\text{Sr}$  relative to  $^{137}\text{Cs}$  as compared to the general global  $^{137}\text{Cs}/^{90}\text{Sr}$  fallout ratio

Table 2. Definitions of water masses (Erik Buch, pers. comm.)

Water Mass		T: Potential temperature, °C S: Salinity, psu.
PW	Polar Water	$S < 34.4$
PIW*	Polar Intermediate Water	$34.4 < S < 34.7, T < 0$
ASW*	Arctic Surface Water	$34.4 < S < 34.7, T > 0$ $34.7 < S < 34.9, T > 2$
AIW	Arctic Intermediate Water	$34.7 < S < 34.9, T < 2$ $S > 34.9, 0 < T < 3$
UPDW	Upper Polar Deep Water	$34.92 < S < 34.92, -0.5 < T < 0$
EBDW	Eurasian Basin Deep Water	$S > 34.92, T < -0.5$
AW	Atlantic Water	$S > 34.9, T > 3$

\*: Data for PIW and ASW have been included with the AIW

Table 3: September 1988 Cruise. Average radionuclide concentrations in different water masses: mean, SD and n (number of results). PW: Polar Water, AIW: Arctic Intermediate Water, UPDW: Upper Polar Deep Water, EBDW: Eurasian Basin Deep Water, AW: Atlantic and Icelandic Coastal water

	$^{99}\text{Tc}$	$^{137}\text{Cs}$	$^{90}\text{Sr}$	$^{137}\text{Cs}/^{90}\text{Sr}$	$^{99}\text{Tc}/^{90}\text{Sr}$	Ch- $^{137}\text{Cs}$ **	Old $^{137}\text{Cs}/^{90}\text{Sr}$
	mBq m <sup>-3</sup> ± SD(n)	Bq m <sup>-3</sup> ± SD(n)	Bq m <sup>-3</sup> ± SD(n)		× 1000	Bq m <sup>-3</sup> ± SD(n)	
All:							
PW	75 ± 24(14)	8.6 ± 0.8(12)	3.7 ± 0.5(12)	2.4 ± 0.2(12)	22 ± 6(12)	1.3 ± 0.3(11)	2.0 ± 0.1(11)
AIW*	45 ± 13(17)	6.2 ± 1.2(17)	2.2 ± 0.4(16)	2.8 ± 0.4(16)	21 ± 5(16)	2.2 ± 0.8(12)	1.9 ± 0.3(11)
UPDW	34 ± 2(3)	3.6 ± 1.0(3)	1.6 ± 0.3(3)	2.3 ± 0.3(3)	22 ± 5(3)	2.3 ± 0.6(2)	1.1 ± 0.2(2)
EBDW	30 (1)	2.3 (1)	0.8 (1)	2.7 (1)	35 (1)	0.5 (1)	2.1 (1)
71°N							
PW	93 ± 14(6)	9.0 ± 0.7(6)	3.8 ± 0.7(6)	2.4 ± 0.2(6)	25 ± 4(6)	1.5 ± 0.1(5)	2.1 ± 0.1(5)
AIW*	45 ± 13(14)	6.4 ± 1.1(14)	2.2 ± 0.4(13)	2.9 ± 0.4(13)	21 ± 4(13)	2.3 ± 0.8(11)	1.9 ± 0.3(10)
DK-Str							
PW	66 ± 23(5)	8.0 ± 0.4(5)	3.5 ± 0.4(5)	2.3 ± 0.2(5)	19 ± 8(5)	1.1 ± 0.4(5)	2.0 ± 0.1(5)
AIW*	47 ± 18(3)	5.5 ± 1.1(3)	2.4 ± 0.6(3)	2.4 ± 0.3(3)	21 ± 11(3)	1.0 (1)	1.8 (1)
AW/IC	16 ± 7(2)	2.9 ± 0.1(2)	1.6 ± 0.1(2)	1.8 ± 0.0(2)	10 ± 4(2)	0.5 (1)	1.4 (1)

\*: PIW and ASW have been included in the AIW

\*\* : Chernobyl  $^{137}\text{Cs}$ , Sept 1988:  $^{134}\text{Cs}$  / 0.26

Table 4: September 1990 Cruise. Average radionuclide concentrations in different water masses: mean, SD and n (number of results). PW: Polar Water, AIW: Arctic Intermediate Water, UPDW: Upper Polar Deep Water, EBDW: Eurasian Basin Deep Water, AW: Atlantic and Icelandic Coastal water

	$^{99}\text{Tc}$ mBq m <sup>-3</sup> ±SD(n)	$^{137}\text{Cs}$ Bq m <sup>-3</sup> ±SD(n)	$^{90}\text{Sr}$ Bq m <sup>-3</sup> ±SD(n)	$^{137}\text{Cs}/^{90}\text{Sr}$	$^{99}\text{Tc}/^{90}\text{Sr}$ × 1000
All:					
PW	70 ±25(10)	7.3 ±1.8(12)	3.1 ±0.7(12)	2.4 ±0.3(12)	21 ±6(8)
AIW*	35 ±10(15)	5.0 ±0.7(17)	1.9 ±0.3(17)	2.7 ±0.4(17)	20 ±6(13)
UPDW	36 ±5(3)	4.6 ±1.5(2)	1.8 ±0.4(2)	2.6 ±0.3(2)	19 ±3(2)
EBDW	18 ±2(5)	1.9 ±0.7(6)	0.8 ±0.1(6)	2.2 ±0.4(6)	21 ±3(5)
71°N					
PW	62 ±17(5)	7.7 ±1.4(4)	3.1 ±0.6(4)	2.5 ±0.3(4)	20 ±3(4)
AIW*	36 ±12(11)	4.8 ±0.6(10)	1.8 ±0.2(10)	2.7 ±0.2(10)	22 ±7(9)
DK-Str					
PW	98 ±7(3)	8.6 ±0.5(5)	3.5 ±0.4(5)	2.5 P0.4(5)	27 ±2(3)
AIW*	32 ±4(4)	5.2 ±0.9(7)	1.9 ±0.4(7)	2.8 ±0.5(7)	17 ±5(4)
AW/IC	8 (1)	2.3 ±0.1(2)	1.7 ±0.1(2)	1.3 ±0.0(2)	4.4 (1)

\*: PIW and ASW have been included in the AIW

of  $\sim 1.5$ . This excess  $^{90}\text{Sr}$  probably originated from riverine sources, where caesium is kept back relative to strontium, e.g. the Ob river system draining contaminated nuclear weapons production areas in the Urals.

Another large contributor to North East Atlantic anthropogenic radioactivity is controlled releases from European nuclear fuel reprocessing plants, especially Sellafield (formerly Windscale) in UK discharging to the Irish Sea. Throughout the 1980's, the Sellafield contribution has gradually increased the  $^{137}\text{Cs}/^{90}\text{Sr}$  ratio in the North East Atlantic including the East Greenland Current (Aarkrog et al. 1983, 1987, Livingston 1985, 1988). For  $^{99}\text{Tc}$ , the release pattern is different with relatively high releases 1970-1980, and lower releases 1981-1986. During 1978-1984, which is relevant for the present data, the annual discharges from Sellafield showed average  $^{99}\text{Tc} / ^{137}\text{Cs}$  ratios of  $0.014 \pm 0.015(\text{SD})$ , which is two orders of magnitude higher than the theoretical fission ratio  $1.44 \times 10^{-4}$  (Aarkrog et al., 1986) assumed to be present in global fallout.

Finally, the Chernobyl Nuclear Power plant accident April/May 1986 gave significant injections of  $^{137}\text{Cs}$  and  $^{134}\text{Cs}$ , especially in the Baltic Sea but also at various locations throughout the North East Atlantic. Relatively high levels have been seen in the Irish Sea, the southern North Sea and the Norwegian Coastal Current, whereas the contributions directly to the Barents Sea and the Arctic Ocean are largely unknown, although they are thought to be low. Measurements in 1987, one year after the accident, showed a pronounced contamination in the Norwegian Coastal Current with gradually decreasing levels over the Norwegian Sea and the Greenland Sea towards the Greenland Coast. The levels in the surface Polar Water of the East Greenland Current was very low (Aarkrog et al 1989).

The radionuclide data can help elucidating the origin of some of the water masses. In the following, the observed water masses in the study area will be discussed on the basis of their radionuclide signal.

#### Arctic Intermediate Water (AIW)

The AIW is where the European radionuclide signal is thought to appear first. In this account, the minor water masses PIW and ASW have been included in the AIW. Chernobyl caesium (only 1988 data) is highest in the warm, saline "Return Atlantic Current" core. A scatter plot of the measured values for Chernobyl Caesium in the AIW shows, however, a large variability for the high salinities (Fig. 4). This might indicate different origins for different parts of the high salinity AIW, but probably it should to a large extent be explained by the very inhomogeneous input function. For  $^{90}\text{Sr}$ , total  $^{137}\text{Cs}$  and  $^{99}\text{Tc}$  there is a tendency towards lower activity for higher salinity (Fig. 4). For  $^{137}\text{Cs}$  and  $^{90}\text{Sr}$  a linear regression of activity against salinity gives a significantly negative slope. The observed Chernobyl Caesium is in accordance with a fast transport of the Atlantic core of the AIW from the Norwegian Coastal Current, where high levels of Chernobyl  $^{137}\text{Cs}$  was deposited in 1986. The negative correlation of  $^{90}\text{Sr}$ , total  $^{137}\text{Cs}$  and  $^{99}\text{Tc}$  with salinity in the AIW can be explained by the combined effect of two phenomena: the background level of  $^{137}\text{Cs}$  and  $^{90}\text{Sr}$  from atmospheric fallout and other old sources has since the 1960'es been higher in the Polar Water than in the Atlantic water, and furthermore the Sellafield

discharges of especially  $^{137}\text{Cs}$  were declining sharply during 1980 - 1985, i.e. the period relevant for the two samplings. Probably a few years earlier, e.g. when Sellafield radiocaesium was first observed in the East Greenland Current in 1982 (Aarkrog et al., 1983), the correlation with salinity might have been opposite, although it was not measured at that time. The scatter around the regression lines (Fig. 4) increasing in the order  $^{90}\text{Sr} < ^{137}\text{Cs} < ^{99}\text{Tc} < \text{Chernobyl } ^{137}\text{Cs}$  may be caused by an increasing input variability in the same order: 1) The deposition of Chernobyl  $^{137}\text{Cs}$  took place in a short time span, but was geographically exceedingly scattered; 2) The main source of  $^{99}\text{Tc}$  in the area is Sellafield,  $\sim 85\%$  (cf. Table 5), and the nature of this discharge is very variable with time; 3) The  $^{137}\text{Cs}$  discharge is somewhat less pulsed and furthermore, the Sellafield source gives only a  $\sim 50\%$  contribution; 4) Finally for  $^{90}\text{Sr}$ , only  $\sim 15\%$  comes from the European sources resulting in much more steady levels. However, a minor part of the scatter may be attributed to increasing analytical error terms, which in fact increases in the same order.

#### Upper Polar Deep Water (UPDW)

The concentration of all the observed nuclides in the AIW are significantly different from the levels in the PW and in the EBDW, whereas it cannot be clearly distinguished from the UPDW based on the present radionuclide levels. The relatively high content of Chernobyl  $^{137}\text{Cs}$  in the UPDW (Fig. 2) - as in the AIW - is also noteworthy although it only concerns two samples. Provided these samples have been correctly classified, this points to a similar fast production route as for the AIW, maybe a sinking of West-Spitzbergen water in the Fram Strait, or maybe a Barents Sea shelf source, and a fast return flow through the Fram Strait under the AIW in the East Greenland Current.

#### Eurasian Basin Deep Water (EBDW)

The data for the EBDW only consist of one good data set in 1988, whereas the 1990 sampling gave more results. All measured radionuclides are clearly lower than in the AIW and the UPDW. The level of  $^{137}\text{Cs}$  and  $^{90}\text{Sr}$  in the EBDW in 1990 (Table 4) was in the present study 70% and 45%, respectively, higher than the same water mass observed in the Arctic Ocean 1981-1984 as referred by Swift and Koltermann (1988). Such an increase during transit from the Arctic Ocean could be expected due to a gradual mixing with the overlying UPDW or other water masses carrying higher levels of contamination. The larger increase for  $^{137}\text{Cs}$  as compared to  $^{90}\text{Sr}$  supports this, as  $^{137}\text{Cs}$  from Sellafield has become gradually more important in the period. Furthermore, the radionuclide levels referred by Swift and Koltermann (1988) may not be entirely representative for the part of the EBDW flowing out of the Arctic Ocean to form the presently observed water mass. Thus two independent observations in the Arctic Ocean has shown distinct  $^{137}\text{Cs}$  peaks in the deep water (1500 m) possibly originating from deep water formation on the Arctic Ocean shelf (Livingston et al. 1984, Livingston 1988, Smith et al. 1990). It should furthermore be noted that in spite of relatively few observations, the 1990 EBDW samples from the Jan Mayen section indicates a gradual decrease of  $^{137}\text{Cs}$  and  $^{90}\text{Sr}$  towards the bottom (Fig. 3). This indicates that an internal structure in the EBDW can be distinguished as it was observed for the AIW.

### Polar Water (PW)

In the PW, the Chernobyl signal is clearly smaller and the other nuclides are higher than in the AIW. Furthermore, the non-chernobyl nuclides show highest concentrations in the surface and closest to the Greenland coast - especially in 1988. In 1982, higher concentrations of Sellafield derived caesium was seen in surface water samples from the East Greenland Current further away from the coast (Aarkrog et al. 1983). A transit time relative to Sellafield of 7-8 years for the outer part of the East Greenland Current was then estimated. Based on  $^{99}\text{Tc}$  in seaweed samples from the East Greenland coast, this was confirmed for the coastal part of the East Greenland Current (Aarkrog et al., 1987). As the coastal sites now seems to carry the highest European signal, when the relevant discharge rates are decreasing, we may stipulate a prolonged "tail" effect, i.e. a spreading of a specific discharge over several years probably due to dispersion and different transit routes leading to a spreading of the "transit time" to maybe 7-10 years or more. What is the transit route for the relatively high activity levels in the PW? The observed structure points against an extensive *in situ* mixing with the underlying AIW, that is - at present - transporting lower levels of the European spike. Probably the transit route goes via the Arctic Ocean surface water, where a fraction of the Atlantic Water carrying the activity from Sellafield has been diluted with fresher water masses. This could take place in the river dominated Arctic Ocean shelf seas, e.g. in the Barents Sea and the Kara Sea, before entering the Arctic Ocean. This route would be consistent with a longer transit time for the PW than the AIW, which also support the presently observed higher activity levels in the PW as the presently observed values then refer to a period with higher discharge rates. The lower Chernobyl signal in the PW compared to the AIW is consistent with a lower Chernobyl contamination of the Arctic Ocean relative to the Norwegian Coastal Current.

### Summary of transit times and routes from European radionuclide sources.

The transit time aspect of the North East Atlantic large scale circulation including the subpolar (or "Nordic") seas has been elucidated by two completely independent events: the European coastal discharges of various radionuclides, mainly to the Irish Sea (Livingston et al. 1982 a,b, Aarkrog et al. 1983, Dahlgard et al. 1986, Aarkrog et al 1987, Smith et al. 1990), and the "Great Salinity Anomaly" (Dickson et al. 1988). Relating the transit times found in these studies to the Sellafield (Irish Sea) discharge pipe, gives the following summary (Figure 5): The warm, saline Atlantic Water (AW), branching off from the Norwegian Coastal Current off North Norway (year 4-5 after Sellafield) to form the West Spitzbergen Current, is gradually cooled on its way to the Arctic Ocean. As a result submerged Atlantic water masses are formed in the Fram Strait, from where it flows southwards below the Polar Water (PW) in the East Greenland Current, contributing to the warm, saline "Atlantic Return Water" (Paquette et al., 1985) core of the AIW. The UPDW may have a similar transit route. The AIW and the UPDW are assigned an age relative to the Norwegian Coastal Current of 1-2 years due to the Chernobyl signal being very clearly present in 1988, i.e. 5-6 years relative to Sellafield. The PW, which is transported to the study area from the Arctic Ocean, shows a longer transit time, approximately 7-10 years or more, relative to Sellafield, and a transport route that may go via the Barents Sea and the Kara Sea. Even the EBDW is influenced by European  $^{137}\text{Cs}$  as the  $^{137}\text{Cs}/^{90}\text{Sr}$  ratio is higher than the fallout ratio of  $\sim 1.5$ .

### Inherent structures in the water masses and variations between years

As discussed above there is much more structure in the water masses than appears from the definitions in Table 2. The observed separation of distinct water masses with different contamination levels after thousands of kilometres and years of transport introduces a possibility of error, when too few samples are measured for radionuclides. However, it also indicates the existence of an inherent variability within the water masses (cf. Fig. 4) and it indicates the dominating importance of long distance horizontal advection in the renewal of the water masses in the study area.

### On the radionuclide balance in Polar Water.

Fucus samples from East Greenland sampled 1965 - 1975 showed values of  $\sim 2$  Bq  $^{99}\text{Tc}$   $\text{kg}^{-1}$  dry (Aarkrog et al. 1987) which is equivalent to  $\sim 20$  mBq  $^{99}\text{Tc}$   $\text{m}^{-3}$  in seawater. An effective half life of 12 years was observed for  $^{90}\text{Sr}$  in East Greenland water samples till 1987 (Aarkrog, 1989). Extracting the 29 year physical half life

**Table 5.** Radionuclide balance sheet for East Greenland Polar Water.

	1988	1990	Avg.
Measured $^{99}\text{Tc}$ , mBq $\text{m}^{-3}$	75	70	
European $^{99}\text{Tc}$	65	60	63
Sellafield 7-10 yr, TBq $^{99}\text{Tc}$ $\text{yr}^{-1}$	71	18	
La Hague 5-8 yr, TBq $^{99}\text{Tc}$ $\text{yr}^{-1}$	7	14	
Sum, TBq $\text{yr}^{-1}$	78	32	55
TF, European Tc mBq $\text{m}^{-3}$ /TBq $\text{yr}^{-1}$	0.8	1.9	1.1
Sellafield 7-10 yr, PBq $^{137}\text{Cs}$ $\text{yr}^{-1}$	3.03	2.15	
European $^{137}\text{Cs}$ , Bq $\text{m}^{-3}$ (TF=1.1)	3.4	2.4	
Measured $^{137}\text{Cs}$ , Bq $\text{m}^{-3}$	8.6	7.3	
Chernobyl $^{137}\text{Cs}$ , Bq $\text{m}^{-3}$	1.3	$\sim 1.1$	
Calculated "Old" $^{137}\text{Cs}$ , Bq $\text{m}^{-3}$	3.9	3.8	
"Old" $^{137}\text{Cs}$ , extrapolated	3.6	3.1	
European $^{90}\text{Sr}$ , Bq $\text{m}^{-3}$ (TF=1.1)	0.5	0.4	
Measured $^{90}\text{Sr}$ , Bq $\text{m}^{-3}$	3.7	3.1	
Calculated "Old" $^{90}\text{Sr}$ , Bq $\text{m}^{-3}$	3.2	2.7	
"Old" $^{90}\text{Sr}$ , extrapolated	3.2	2.8	
Extrapolated "old" $^{137}\text{Cs}/^{90}\text{Sr}$	1.1	1.1	
Presently measured $^{137}\text{Cs}/^{90}\text{Sr}$ , PW	2.4	2.4	



gives a 20 year half life for stable strontium, which is probably also applicable for fallout  $^{99}\text{Tc}$ . The background from fallout and possibly other early sources may therefore be estimated as  $10 \text{ mBq } ^{99}\text{Tc m}^{-3}$  in East Greenland PW 1988 - 1990. Table 5 is a "back of an envelope" balance sheet for  $^{99}\text{Tc}$ ,  $^{137}\text{Cs}$  and  $^{90}\text{Sr}$  in East Greenland Polar Water. After subtracting the  $10 \text{ mBq m}^{-3}$  background, the remaining  $^{99}\text{Tc}$  is assumed to originate from the European reprocessing plants. Dividing the European  $^{99}\text{Tc}$  concentration with the sum of the average  $^{99}\text{Tc}$  discharge 7-10 years and 5-8 years earlier, respectively, from Sellafield and La Hague, gives transfer factors  $0.9 - 1.9 \text{ Bq m}^{-3} / \text{PBq yr}^{-1}$  (Table 5) between Europe and the East Greenland PW. The effect of the decreasing discharges has not been fully effectuated in the average levels in the water indicating that there may be an even longer "tail" effect, i.e. due to dispersion and different transit routes, a specific annual discharge will be spread over several years in the study area. The TF value  $1.1 \text{ Bq m}^{-3} / \text{PBq yr}^{-1}$  based on average  $^{99}\text{Tc}$  values between the two samplings have therefore been used to calculate "European"  $^{137}\text{Cs}$  and  $^{90}\text{Sr}$ . Values for the "old"  $^{137}\text{Cs}$  and  $^{90}\text{Sr}$  have then been calculated by subtracting the European contribution and - for  $^{137}\text{Cs}$  - the Chernobyl contribution from the measured concentrations. Based on seawater data from the East Greenland coast in the period 1974 - 1985, regression lines for the assumed "old"  $^{137}\text{Cs}$  and  $^{90}\text{Sr}$  concentrations have been calculated (Aarkrog 1989). Extrapolated values from these regression lines fit remarkably well with the above mentioned calculated "old"  $^{137}\text{Cs}$  and  $^{90}\text{Sr}$  values.

There are large uncertainties and assumptions in the above estimates. One uncertainty is, that the transfer factor from European discharges could be lower for caesium than for  $^{99}\text{Tc}$  due to sedimentation with suspended material, e.g. in the Irish Sea and the North Sea and maybe further in the river water influenced Barents Sea and Kara Sea. That would reduce the European  $^{137}\text{Cs}$  contribution.

However, within reasonable error terms the present data gives no indications of significant additional sources to the studied water masses than the already described European contributions, Chernobyl and "old" sources including global atmospheric fallout and pre-1970'es sources to the Arctic Ocean and its shelf seas. Thus the early Soviet discharges in the 1940'es - 1960'es are included in the "old" sources, and they are probably responsible for the observed higher level of  $^{137}\text{Cs}$  and  $^{90}\text{Sr}$  in the East Greenland Current as compared to open Atlantic Water (Faroe Islands) observed throughout the 1960'es and 1970'es (Aarkrog 1989).

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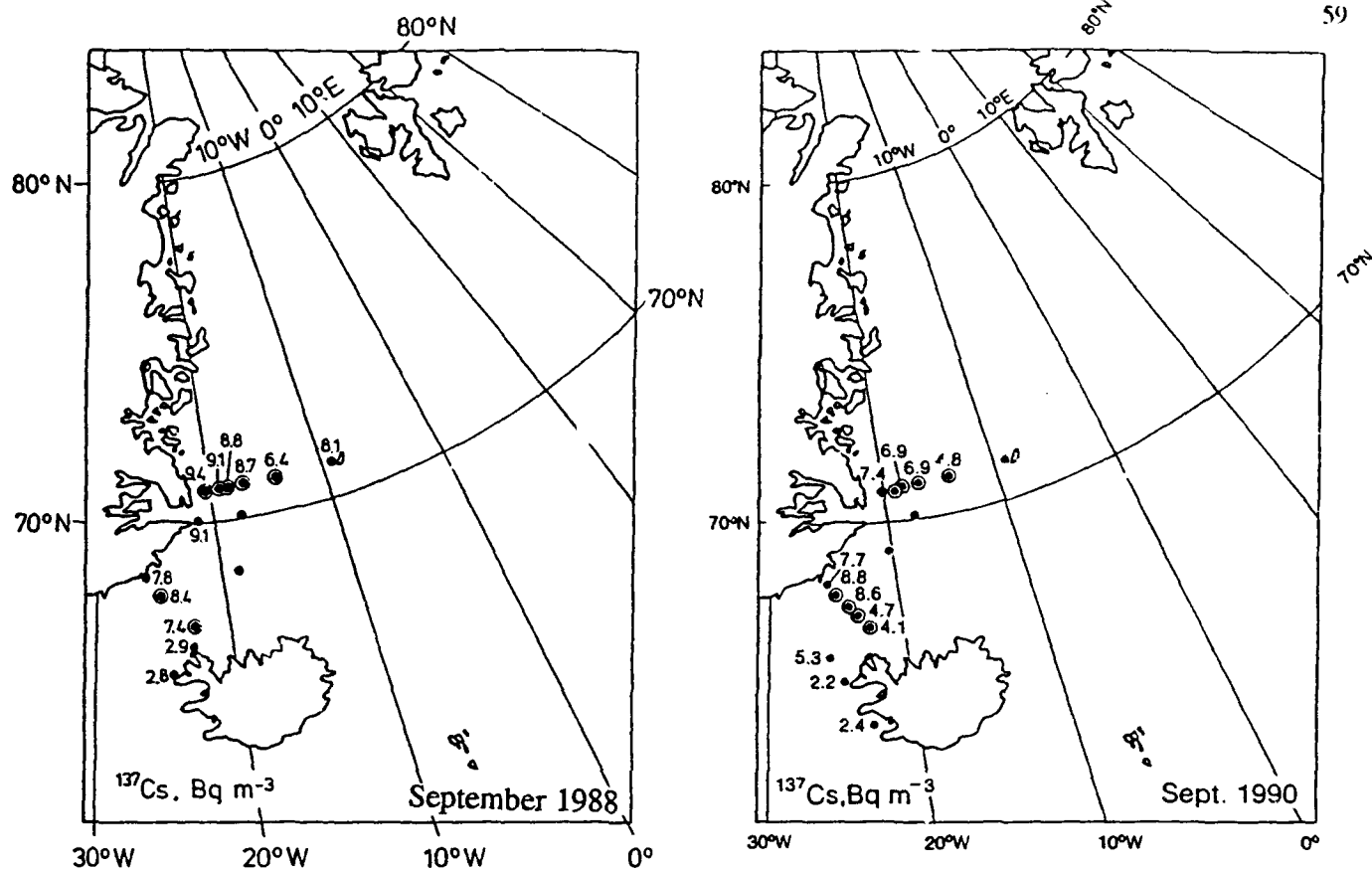


Figure 1. Sampling locations September 1988 (a) and 1990 (b) and  $^{137}\text{Cs}$  in surface water samples.  $\odot$  : Deep profiles, see figures 2 - 3.

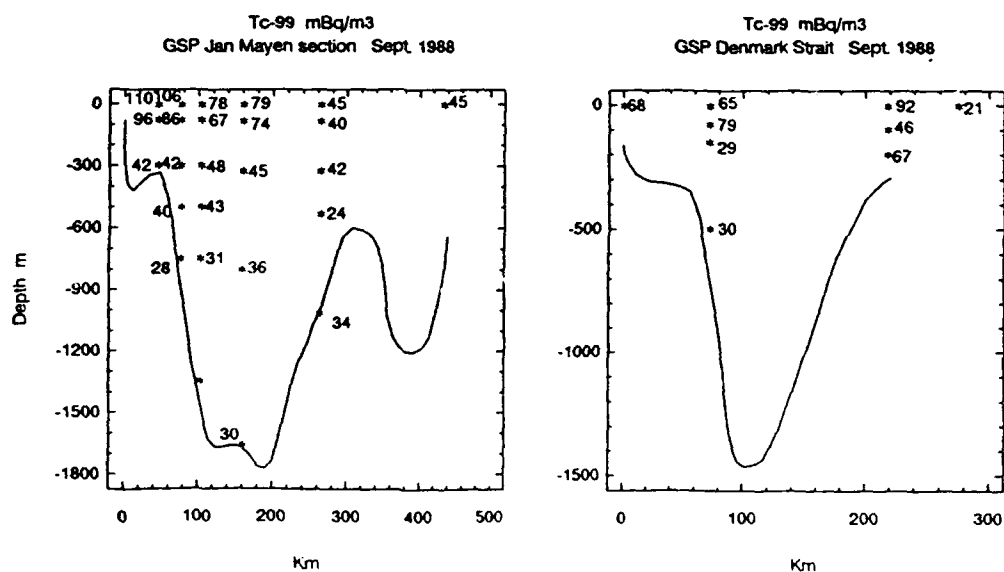
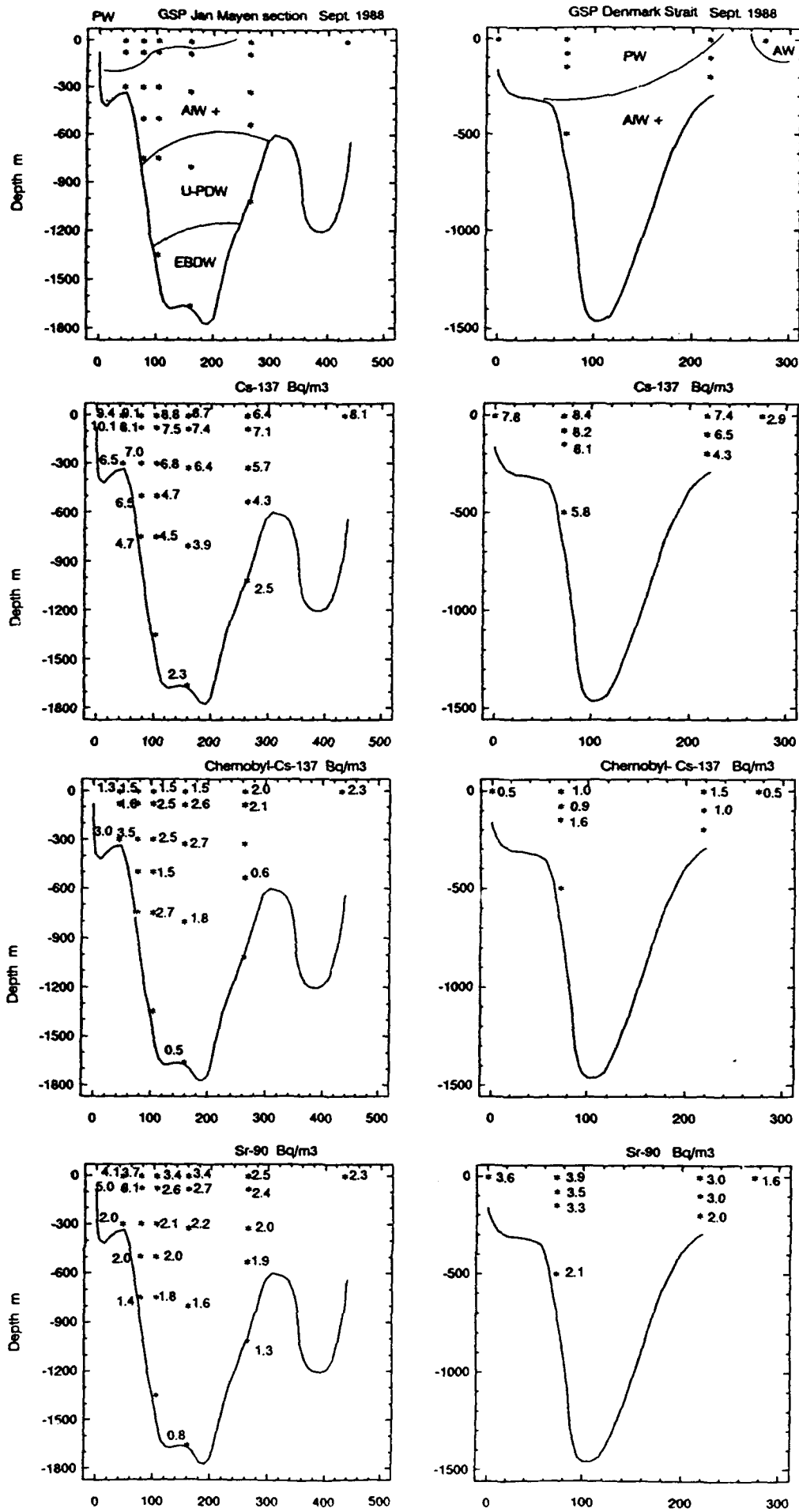


Figure 2, continued.



**Figure 2. Radionuclide data from the two deep water sections September 1988**

**Figure 3. Radionuclide data from the two deep water sections September 1990.**

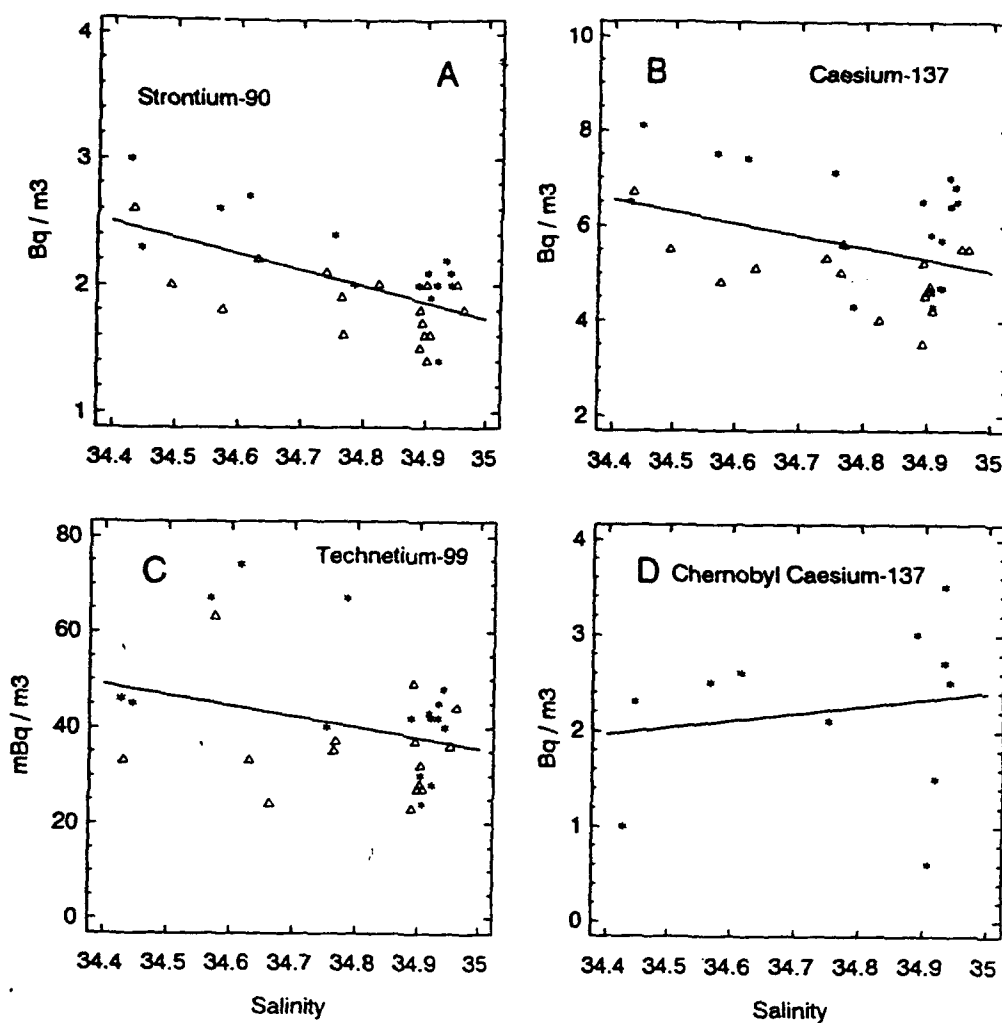


Figure 4. Radioisotope concentrations as a function of salinity and linear regression lines for the AIW. \*: 1988 data,  $\Delta$ : 1990 data. The slopes of the regression lines for  $^{90}\text{Sr}$  and  $^{137}\text{Cs}$  are significantly different from zero.

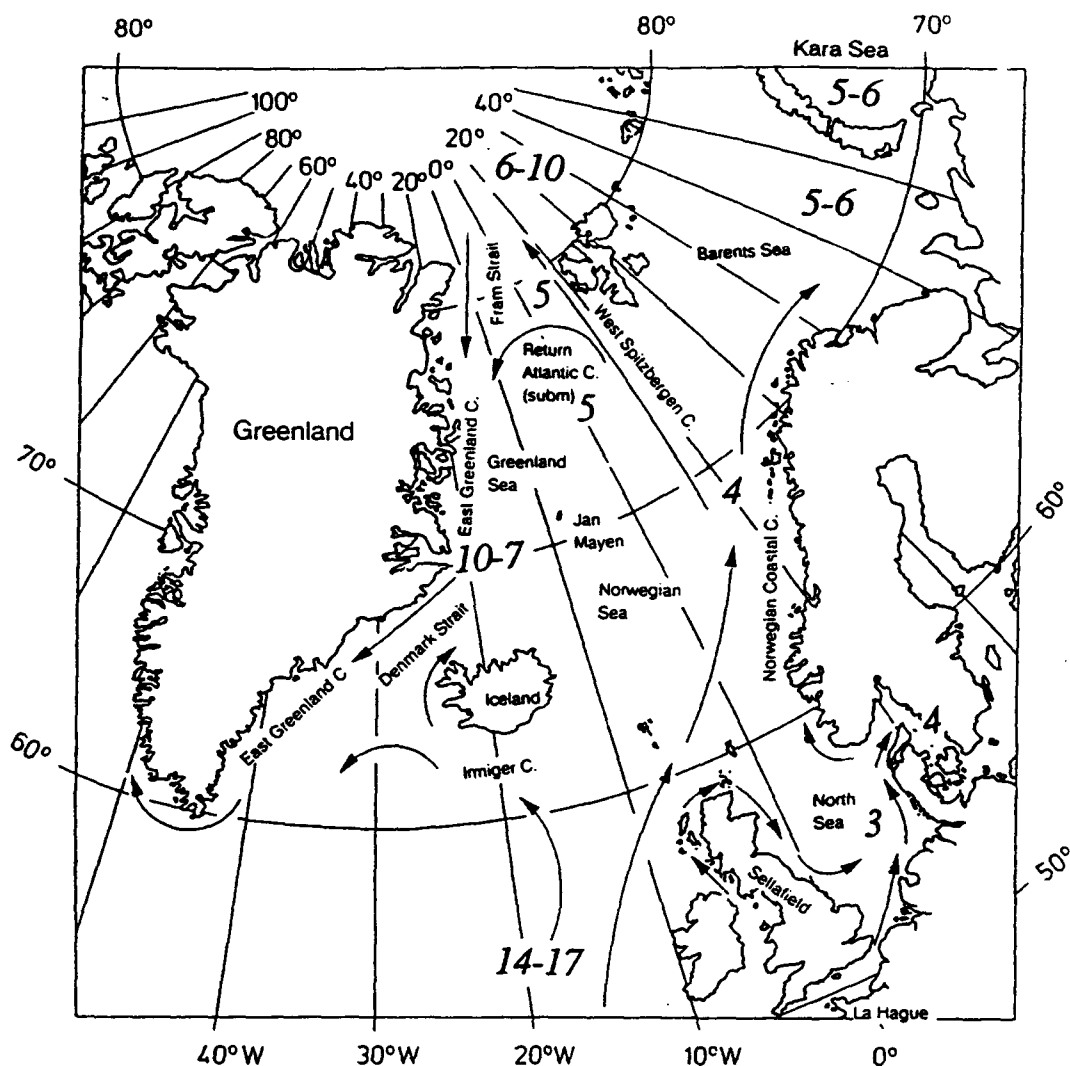


Figure 5. Major surface currents and transit times in years from Sellafield to different sea areas. The indicated transit time (14 - 17 years) for the North Atlantic current is deduced from Dickson et al. (1988) relative to the East Greenland Current PW. The La Hague discharge will be 2 years ahead.





## **RADIOACTIVITY AND ENVIRONMENTAL PROBLEMS FOR THE SEAS AND OCEAN**

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The pollution of the world ocean and disturbance of major-biogeochemical cycles have acquired a global nature observed in the recent thirty years and resulted in the occurrence of global ecological problems, up to the mass disturbance in marine ecosystems and global climate change. Particularly susceptible is the system of some polar seas.

The world ocean occupies the terminal chain of large-scale irreversible fluxes of various substances, which is of a great importance in the existence of all living things of the Planet [1].

Of a particular danger is the ocean pollution by xenobiotics, and in the atomic era—by anthropogenic radionuclides.

The atmospheric transport and continental runoff are major sources of the ocean pollution by hazardous anthropogenic pollutants.

The radioactive sources of the world ocean pollution can be classified in more detail:

1. Contamination from nuclear weapon tests—global contamination from atmospheric tests—primarily before 1963, and local contamination from tests conducted directly in the sea or coastal zone, though they were not numerous.
2. Radioactive dumping from radiochemical plants directly into the sea or rivers.
3. Radioactive waste contamination (solid wastes which are of potential danger, and liquid wastes dumped directly into the sea).
4. Large-scale accidents leading to direct or indirect entering of the radioactivity into the sea or ocean (e.g., the chernobyl accident, ship and space type nuclear plant accidents, etc.).

5. Other less sizable contamination. During nuclear weapon tests, particularly before 1963, a large amount of radionuclides were emitted into the atmosphere; long-lived explosive products deposited gradually over the earth surface. The total content of artificial radionuclides in the world ocean before 1970 was 200–600 MCi (Preston, 1974), and additionally of tritium—1000 MCi. The extensive investigation of the Pacific and Atlantic water radioactivity was carried out in 1964–67, just after the ceasing of the atmospheric weapon tests, (ref. Fig. 1), and the results are given in paper /2/.

The highest radioactive concentrations are located in regions of the tests in the Pacific ocean (Marshall Islands). On a whole the contamination observed in the Pacific was much higher than that in the Atlantic (ref. Fig. 2).

Most dangerous were radionuclides of caesium-137 and strontium-90, as well as cerium-144, yttrium-91 and niobium-95.

The radioactive concentrations in the Pacific in 1965–70 dropped gradually while in the Atlantic they remained at a constant level, and in 70s (latitude) of the Northern Hemisphere they usually amounted to 2.5–8 Bq/m<sup>3</sup>, and in 40s N—from 5 to 25 Bq/m<sup>3</sup>.

Increased concentrations of strontium-90 and caesium-137 were detected in 1964 and 1967 in the Pacific region where the Columbia river (US) falls into the ocean /2/. The mean concentrations here were: Sr-90—13–18 Bq/m(3), caesium-137—17 Bq/m(3). The radioactivity store has been determined here in the 1000 m water column. The obtained values of 100–420 mCi/sq.m for strontium-90 and 180–340 mCi/sq.m for caesium-137 are completely outside the measurement results obtained over the rest water area of the Pacific. The comparison with the data on the accumulated Sr-90 fallouts indicates that the global radioactive fall-out amount to only 20% of the store observed. The strontium-90 store in the ocean sector with a radius of 435 km, area of 220 thousand sq.km and a center located in the Columbia river mouth is estimated as 70 kCi and that of caesium-137—100 kCi.

This contamination was apparently attributed at that time to radioactive disposals from Handford plants. It is emphasized in paper /3/ that radioisotopes, which enter the Columbia river with cooling Handford reactor waters, are transported in the ocean at a distance up to 650 km. A substantial migration of radionuclides in the marine water which is much related to their state (forms) has been confirmed /2/.

It is noteworthy, that the living substance, which removes permanently chemical elements from the marine water and incorporates them into the biocirculation system is one of the components of the oceanic suspension. Iron is another significant component of the oceanic suspension. While producing hydroxide it can promote the extensive extraction of elements from the marine environment. The entrapment of many elements by the suspension is above a factor of 10 /3/.

It is found that the accumulation factor of all isotopes, when the iron is available, by the plankton is of an order of a magnitude higher than that in the marine water-planet-isotope system.

The radioelements when being entrapped by the evermore new plankton organisms are transported from protozoan to higher organisms directly threatening them and then man despite the available notion about the infinite dilution of radioelements in the ocean.

The data obtained are indicative of the danger produced by radioactive elements disposed into the ocean. I wish to emphasize that the conclusion was already made 25 years ago.

Let me dwell in detail on the radioactive contamination of Arctic Seas washing the territory of Russia.

Six Arctic Seas wash its territory: White Sea, Barents Sea, Kara Sea, East Siberian Sea, Laptev Sea and Chukchi Sea. The total sea surface area of these seas is 4.6 million square kilometers, which is 31% of the total area of the Arctic ocean. These seas were mainly contaminated by the global radioactive fallout from atmospheric nuclear weapon tests; radioactive products transported by river runoffs; by radioactive wastes from the radiochemical plant in Sellafield (UK) and C. La Hague (France) disposed into the Ireland Sea and brought into the Barents Sea by the Gulf Stream; unsafe vessels and nuclear energy plants sunk in the bays of the island Novaya Zemlya. Table 1 gives estimates of radioactive products entered the Arctic Seas based on our investigations and literature data /5,6/. One should mention here about nuclear weapon tests at site Novaya Zemlya. In accordance with data in book /7/ from 132 nuclear explosions conducted at the polar site 87 explosions were conducted in the atmosphere: 83 of them were conducted in the air (at a height of 0.7-10 km), 1 was a ground explosion (07.09.57), 3 over the sea; 3 underwater explosions (in "Chernaya" Bay on Novaya Zemlya) (21.09.55, 10.10.57 and 23.10.61) and 43 underground nuclear explosions.

Table 1.

**Sources of Anthropogenic Radionuclides in Arctic Seas  
Washing the Territory of Russia**

Technogenic Radionuclide Source	Quantity of Radioactive Products (kCi)	
	137-caesium 90-strontium	Total beta-activity
1. Global fallout	320	
2. River runoff	100	
3. Income with Gulf Stream	200	
4. Disposal of solid and liquid wastes		30
5. Sunken reactors		325*-2500**
<b>TOTAL</b>	<b>520</b>	<b>355-2530</b>

\*from literature data

\*\*expert estimate of the upper limit of the activity at the moment of burial  
1965-1988.

The atmospheric nuclear bursts must have contributed substantially to the global radioactive fallout. The rest of the explosions, 1 ground and 3 underwater explosions, produce evidently local contamination and underground explosions do not produce significant contamination.

Of all sources listed in the Table, the first four are sources of radioactive contamination of the sea; the 5th source is potential since the sunken reactor compartments with spent nuclear fuel, before being sunk, had been filled with a solidifiable mixture, which, according to estimates made by designers of nuclear energy plants, should prevent the contact of the spent nuclear fuel with the marine water in terms of several hundred years /4/.

Since 1961 the USSR Hydromet service had been providing monitoring of the radioactive contamination of the Arctic Seas. In 1961 observations were made of the total beta-activity of the marine

water. In 1962-63 the content of strontium-90 was started to be observed on a regular basis. The content of caesium-137 in the marine water and sediments were measured only in White, Barents and Kara Seas during special expedition surveys with the periodicity of 3-5 years. In the whole observation period the maximum levels of contamination by strontium-90 were recorded in 1963 and they varied from 0.8 pCi/l (the White Sea) to 0.4 pCi/l (Chukchi Sea). The maximum levels of contamination by  $^{137}\text{Cs}$  of 0.9 pCi/l were recorded in 1982 in the western Barents Sea which received water from the Gulf Stream (Fig. 3). This level was 6 times as much as that of the global water contamination in the North Atlantic. The marine water of that region contained  $^{134}\text{Cs}$  not detected in global radioactive fallouts. In accordance with the expedition survey results radioactive wastes disposed into the Irish Sea and transported by the sea current system into the North Sea and further on along the Norwegian coasts and into Barents, White Kara Seas formed a source of contamination additional to global caesium-137 /5,8/ (Fig. 4). More than 1000 kCi of  $^{137}\text{Cs}$  were disposed into the Irish Sea in the period from 1970 to 1984. Of this 200 kCi entered the Barents Sea /5,9/. This amount of caesium-137 is above that entering the Baltic Sea (70 kCi) and the Black Sea (45 kCi) after the Chernobyl nuclear power plant accident /10/. The annual disposal of caesium-137 into the Irish Sea decreased in the period 1975-1984 from 140 kCi to 12 kCi (in a year), and this resulted in a significant decrease of the caesium-137 concentration in the Barents Sea. In the period 1982-1992 the concentration of caesium-137 in the Barents Sea dropped down to 0.1-0.16 pCi/l, i.e., 5-10 times. The rates of the drop of the concentration of strontium-90 in the marine water, produced primarily by global fallouts, are lower than those of  $^{137}\text{Cs}$ . For the 29-year observation period (1963-1992), the concentration of  $^{90}\text{Sr}$  in the White and Barents Seas decreased only 3-5 times.

Of a particular concern is the information on a possible impact of radioactive products concentrated in the unsafe reactors buried in bays of island Novaya Zemlya on the radiation situation in the Kara Sea. With that end in mind the joint Russian-Norwegian expedition studied the Kara Sea water area (except the Novaya Zemlya Island bays). The observation points nearest to the bays were at a distance of 50-100 km. Based on the preliminary data obtained by the moment the conducted survey according to its state-of-art by summer 1992 did not find any effects of buried radioactive products on the radiation situation in the Kara Sea.

However, recently published data are known about the amount of radionuclides buried in the Kara Sea, and the final issue concerning the seal failure of buried blocks and a possible spread of radionuclides requires a thorough studying.

The radioactive wastes buried in the Kara Sea require a particular investigation /4/.

For comparison let us note that the total amount of the global radioactive fallout ( $^{137}\text{Cs} + ^{90}\text{Sr}$ ) deposited over the World Ocean is estimated as 15 thousand kCi, and the Sellafield plant disposals into the sea—1000 kCi /9/.

The investigations made by joint Soviet-US expedition III (in 1988) demonstrated that the content of  $^{137}\text{Cs}$  in the Bering and Chukchi Sea water over the whole region under survey was, after the averaging, 0.06 pCi/l. The concentrations ranged from 0.04 to 0.1 pCi/l. The maximum caesium-137 concentrations in the Bering Sea were recorded in the 0-40 m layer in the south-west from island Saint Lawrence. The monotonous increase of the caesium-137 concentrations were observed from surface to bottom practically at all observation stations. The maximum concentration gradient was noted in the western Chukchi Sea: caesium-137 concentration in the 0-3 m surface layer was 0.03 pCi/l, at a depth of 40 m it was around 0.1 pCi/l. maximum caesium-137 concentrations are observed in the near bottom layers, the average value is 0.08 pCi/l.

The values of these concentrations are characteristic of the background regions in the world ocean and are related to the global income of  $^{137}\text{Cs}$  from the atmosphere in a long-term period /11/.

The intercomparison of results obtained and data from other regions of the World Ocean indicates that mean caesium-137 concentrations in the Chukchi and Bering Seas were 10-50 times as little as those in the Black, Barents, Baltic and Greenland Seas [11] susceptible to the impact of local radioactive contamination sources.

#### **Ecological Problems of Radioactive and Chemical Contamination of the Marine Environment**

The potential danger of the above given levels of the radioactive contamination is estimated nowadays, based on the available level of our knowledge, with models developed with reference to the radioecological factor.

The IAEA scheme of critical ways how to evaluate the removal of radioactive products can be applied to all types of radioactive and chemical pollution and is of interest.

Modern notions about ecological implications of the ocean pollution are lately just in the way to be formed, and at the same time one reviews fundamentally the possibilities of ecological reserves, adaptation capacity of the biotic component, and "inexhaustibility" of biological resources.

In accordance with the generalized scheme (we have developed) on ecological consequences of the ocean pollution, while considering negative biological consequences of the marine environment pollution we account the response of the organism, population and community when interfering to functioning processes of "a living substance" at cellular, population-biocenotic, and ecosystem levels.

One takes into account the specificity of the oceanic environment most susceptible to long-term impacts of low doses resulting in a gradual accumulation of pollutants and finally in the degradation of the ecosystem.

The developed concept of the ocean assimilative capacity provides an interrelation of fundamental and applied investigations and is aimed at studying and assessing quantitatively all processes which determine the activity of the "removal" and transformation of pollutants in the marine environment. According to this concept any marine ecosystem occupies a limited volume which can be accurately set, and consequently, the assimilative capacity of each specific ecosystem is also a finite quantity which provides an objective characteristic of the available property of the marine environment.

One can specify two aspects when developing the assimilative capacity of the ocean: theoretical and applied. A number of domestic and foreign scientists have been dealing with studying these aspects, in particular the applied one. The idea on the availability of some natural capacity without using the term "assimilative capacity", has been suggested earlier when regulating radioactive impacts on biological objects. This approach was in particular published in 1976 in the IAEA technical report "The effects of the ionizing radiation on aquatic organisms and ecosystems". The results of these investigations have been generalized by Preston. It is noteworthy that issues concerning establishing standards, reducing disposals of contaminants have been considered earlier with account for primarily only a factor of pollution and dispersion and partially chemical transformation. "The tremendous manifestations of living organisms" in

the marine water determine the passing of biogeochemical cycles both of vital elements and pollutants. This prevailed value of a biotic element in the decomposition and removal of chemical toxicants is an inalienable property of normal marine ecosystem functioning.

The conceptual model we have developed for assessing the assimilative capacity of the marine ecosystem includes three major units (Fig. 5):

1. Calculation of mass balances and "life" time of pollutants in the ecosystem.
2. Analysis of the biotic balance in the ecosystem to select the ecological "target".
3. Assessment of "critical" concentrations of pollution impacts (or ecologically permissible impacts) on the ecological target.

The proposed model for assessing the assimilative capacity is realized in the Baltic Sea and is improved for quantitative solutions on artificial pollutants—polychlorinated biphenyls (PCB), the group of chlorinated hydrocarbons, which endanger mostly the life of chemical compounds and are widely spread in the world ocean.

The proposed generalized scheme on ecological consequences of the ocean pollution is the conceptual model for assessing the assimilative capacity of the marine ecosystem and can be applied to the radioecological factor as well.

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### CAPTIONS TO FIGURES

Fig. 1. Strontium-90 concentrations in surface waters of the Pacific in 1966-67 (the circle diameter is proportional to the concentration value).

Fig. 1b. Strontium-90 concentration in surface waters of the Atlantic in 1967 (the circle diameter is proportional to the concentration value)

Fig. 2. Variations in the strontium-90 concentration in surface waters of the northern Pacific and in the Atlantic in 1957-1966.

Fig. 3. Location of sampling stations and distribution of the caesium-137 concentration in surface waters (the circle diameter is proportional to the concentration). Stations in standard hydrological cross-sections have their number given on the left, and the number of the cross-section is given on the right.

Fig. 4. The diagram of the prevailed currents in the Barents Sea. Direction and velocity of the currents (in knots) are shown with arrows.

Fig. 5. Determination of the assimilative capacity of the marine ecosystems.

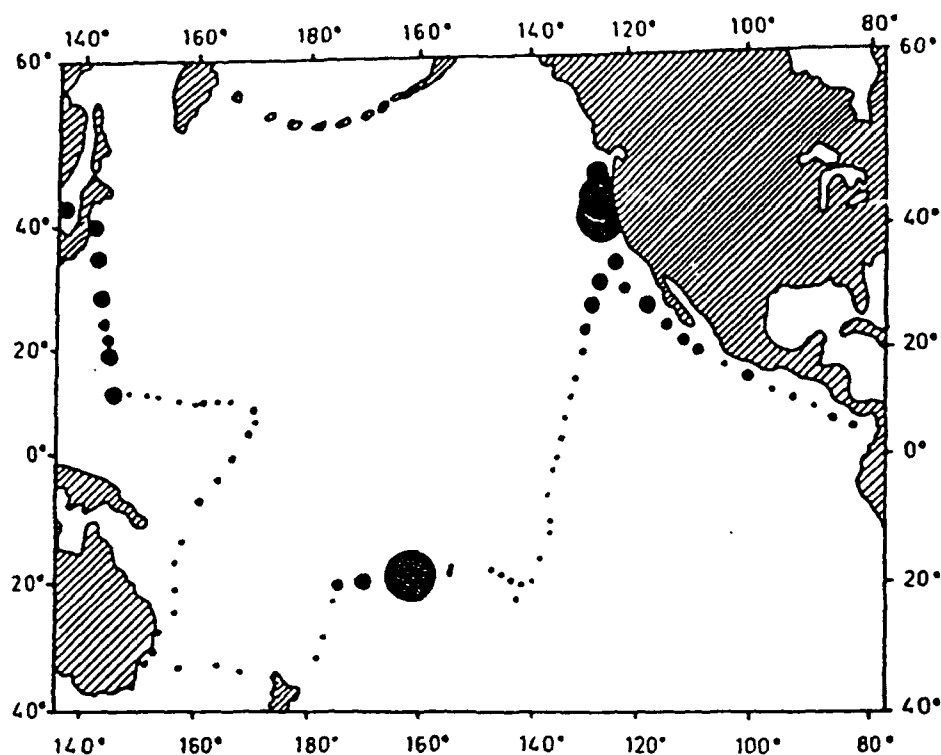


fig Рис. 1а Концентрация  $^{90}\text{Sr}$  в поверхностных водах Тихого океана в 1966-1967 годах (диаметр кружков пропорционален величине концентрации).

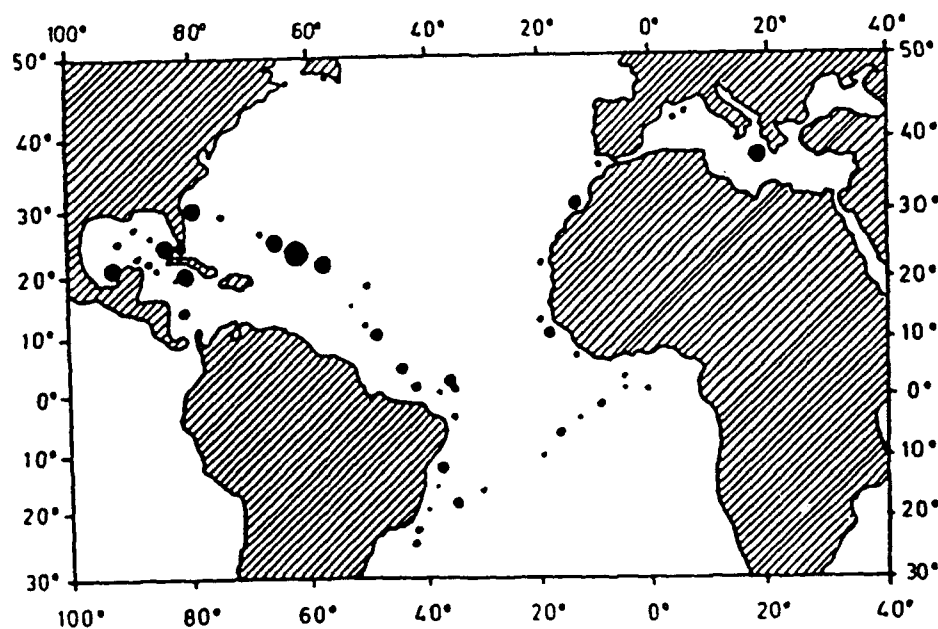


fig Рис. 1б Концентрация  $^{90}\text{Sr}$  в поверхностных водах Атлантического океана в 1967 году (диаметр кружков пропорционален величине концентрации).

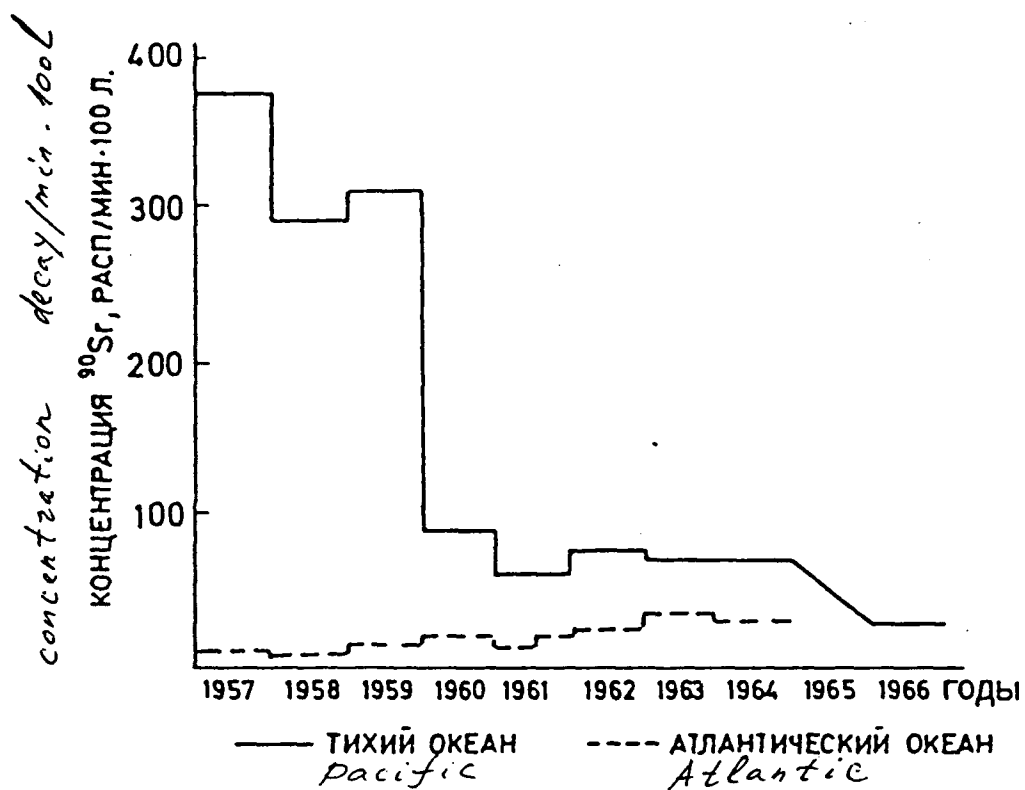


fig Рис. 2. Изменение концентрации  $^{90}\text{Sr}$  в поверхностных водах северной части Тихого океана и в Атлантическом океане за 1957-1966 годы.



## INITIAL DATA

## CHARACTERISTIC TO BE DETERMINED

## CRITERIA FOR THE ASSESSMENT OF ADVERSE EFFECTS

78

Concentration  
of pollutants  
in ecosystem  
components

Biosedimentation  
rate

Pollutant fluxes  
at the borders  
of ecosystems

Rate of microbial  
degradation of  
pollutants

Mass balances  
and the residence  
time of pollutants  
in the ecosystem

Balance disturbance  
and a change in the  
residence time of  
chemical elements  
in the ecosystem

Production/  
destruction  
processes

Community  
structure

The ecological  
"target"

Adverse biological  
effects at the level  
of populations and  
communities

Impact of  
pollutants on  
the basic  
biological  
indices

"Critical"  
concentrations  
of the impact  
of pollutants  
on the ecological  
target

Excess of the  
"critical" concentr-  
ations of pollutants  
in the marine  
environment

Value of the assimilative capacity of  
a marine ecosystem for pollutants

## **RADIOECOLOGY OF THE BARENTS AND KARA SEA: LEVEL OF APPREHENSION AND RESEARCH STRATEGY**

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The Barents and Kara Sea with adjacent territories contain several potential sources of artificial radioactive pollution of regional and global level. The major sources include nuclear weapon tests on Novaya Zemlya and dumping of radioactive waste and used reactors (mainly in the Kara Sea). Radioactive contaminants are also imported to the Barents and Kara Seas by the Gulf Stream and by Ob and Enisey rivers.

During the last two years complex ecological studies of the Arctic continental shelf are being performed by Research Institute for Geology and Mineral Resources of the Ocean (VNII Okeangeologia). Two specialized cruises, aimed at the assessment of radioactive and other types of pollution of bottom sediments, waters, and benthic organisms, have been accomplished in 1991 and 1992 in the Barents and Kara Seas (Figs 1 & 2).

**The main objectives of radioecological study were:**

- estimation of regional background of alpha, beta and gamma emission, with special attention to the Novaya Zemlya shelf;
- investigation of spatial variations in distribution of radionuclides in bottom waters, sediments, and benthic organisms;
- assessment of radiation impact on benthic communities, and measurement of radionuclide concentrations in various trophic groups.

Shipboard studies included towed gamma-spectrometer profiling and on-board measurements of alpha, beta, and gamma emission in the samples of bottom waters and sediments. Detailed measurements of radionuclide activities ( $^{137}\text{Cs}$ ,  $^{134}\text{Cs}$ ,  $^{90}\text{Sr}$ ,  $^{40}\text{K}$ ,  $^{228}\text{Th}$ ,  $^{226}\text{Ra}$ ) are being performed in the laboratories of Arctic & Antarctic Research Institute and Radiation Hygiene Institute. Some of the obtained results are displayed at Figs. 3-5 and Tables 1 & 2.

Table 1. Concentration of radionuclides (Bq/Kg) and total beta activity in the Barents and Kara Sea bottom sediments

Value	$^{137}\text{Cs}$	$^{134}\text{Cs}$	$^{90}\text{Sr}$	$^{228}\text{Th}$	$^{226}\text{Ra}$	$^{40}\text{K}$	Beta activity
1991 cruise data (n=179)							
Mean	2.2			19.5	10.4	400	
Standard dev.	0.9			5.5	3.5	160	
Min	0.0			5.0	2.0	86	
Max	4.0			34.0	19.0	811	
1992 cruise data (n=165)							
Mean	3.4	not detected	not detected	15	9	350	0.702
Standard dev.	0.8			3	5	57	0.112
Min	1.9			10	3	227	0.223
Max	5.6			23	21	453	0.880



**Table 2. Concentration of radionuclides (Bq/Kg) and total beta activity in various sediment texture types**

<b>Sediment texture</b>	<b>Value</b>	<b><math>^{137}\text{Cs}</math></b>	<b><math>^{228}\text{Th}</math></b>	<b><math>^{226}\text{Ra}</math></b>	<b><math>^{40}\text{K}</math></b>	<b>Beta activity</b>
<b>Pure clay (n=84)</b>	<b>Mean</b>	<b>3.49</b>	<b>16</b>	<b>9</b>	<b>351</b>	<b>0.719</b>
	<b>Standard dev.</b>	<b>0.70</b>	<b>4</b>	<b>4</b>	<b>51</b>	<b>0.095</b>
<b>Clay (n=57)</b>	<b>Mean</b>	<b>3.31</b>	<b>15</b>	<b>11</b>	<b>354</b>	<b>0.721</b>
	<b>Standard dev.</b>	<b>0.74</b>	<b>4</b>	<b>14</b>	<b>53</b>	<b>0.098</b>
<b>Silty clay (n=20)</b>	<b>Mean</b>	<b>3.37</b>	<b>15</b>	<b>10</b>	<b>329</b>	<b>0.672</b>
	<b>Standard dev.</b>	<b>0.80</b>	<b>3</b>	<b>5</b>	<b>62</b>	<b>0.113</b>
<b>Clayey silt (n=22)</b>	<b>Mean</b>	<b>3.26</b>	<b>14</b>	<b>8</b>	<b>351</b>	<b>0.681</b>
	<b>Standard dev.</b>	<b>0.76</b>	<b>3</b>	<b>4</b>	<b>53</b>	<b>0.148</b>
<b>Sandy silt (n=10)</b>	<b>Mean</b>	<b>3.58</b>	<b>15</b>	<b>8</b>	<b>369</b>	<b>0.665</b>
	<b>Standard dev.</b>	<b>1.24</b>	<b>5</b>	<b>3</b>	<b>48</b>	<b>0.241</b>

### Preliminary conclusions:

1. Regional background of artificial radionuclide concentrations in bottom sediments is characterized by low levels of  $^{137}\text{Cs}$  (typically  $<5 \text{ Bq/Kg}$ ); no  $^{134}\text{Cs}$  or  $^{90}\text{Sr}$  were detected. Slight anomalies of  $^{137}\text{Cs}$  distribution are revealed in the deep areas west and east of Novaya Zemlya, and north of the Kola Peninsula (Fig. 3).

2. Observed anomalies of natural radionuclide concentrations ( $^{40}\text{K}$ ,  $^{228}\text{Th}$ ,  $^{226}\text{Ra}$ ) are not high enough to produce radioactive stresses on the environment. Areas of increased contents of  $^{226}\text{Ra}$  and/or  $^{228}\text{Th}$  are indicated in the vicinities of the Kolguev Island, Yamal and Taymyr Peninsulas, and in the Ob River estuary (Figs 4 & 5); these probably result from the erosion of granitoid intrusions.

3. Good correlation between concentration of  $^{40}\text{K}$  and total beta-activity ( $r=0.78$ ,  $n=165$ ) suggests that the latter is mostly controlled by this radionuclide.

4. No correlation between radionuclide concentrations and the texture of sediments was revealed (Table 2); yet, no sands were analyzed. The possible link between radionuclides and mineral composition of sediments is under study.

### Future research

Despite the low regional radioactivity level, "hot" local anomalies cannot be ruled out, particularly around the dumping sites of radioactive waste. Investigation of these sites is planned to be carried out in 1993-94. The cruises will be performed on r/v "Geolog Fersman" (Appendix 1). The methodology of research includes:

- identification and observation of dumped objects by means of side-scan sonar and echo-sounder (ship's hull mounted O.R.E. system), as well as sea-floor photography and TV monitoring;
- towed high-sensitivity NaI gamma-spectrometer profiling;
- hydrological measurements (Neil Brown CTD System);
- sampling of water, suspended matter, surficial and downcore bottom sediments, and benthic organisms;
- concentration of large-volume water samples by selective radionuclide sorbents;
- on-board measurements of sample radioactivity with subsequent detailed analysis in the laboratories of Russia, USA, and Canada.

Another research target is the investigation of areas with high sedimentation rates to monitor the impact of radioactive and other types of pollution on the sea-floor environments during the historic time.

Transregional character of oceanographic processes in the Arctic require international co-operation in radioecological studies. We propose the scientific and logistical base of "Okeangeologia" to be used in relevant international projects.

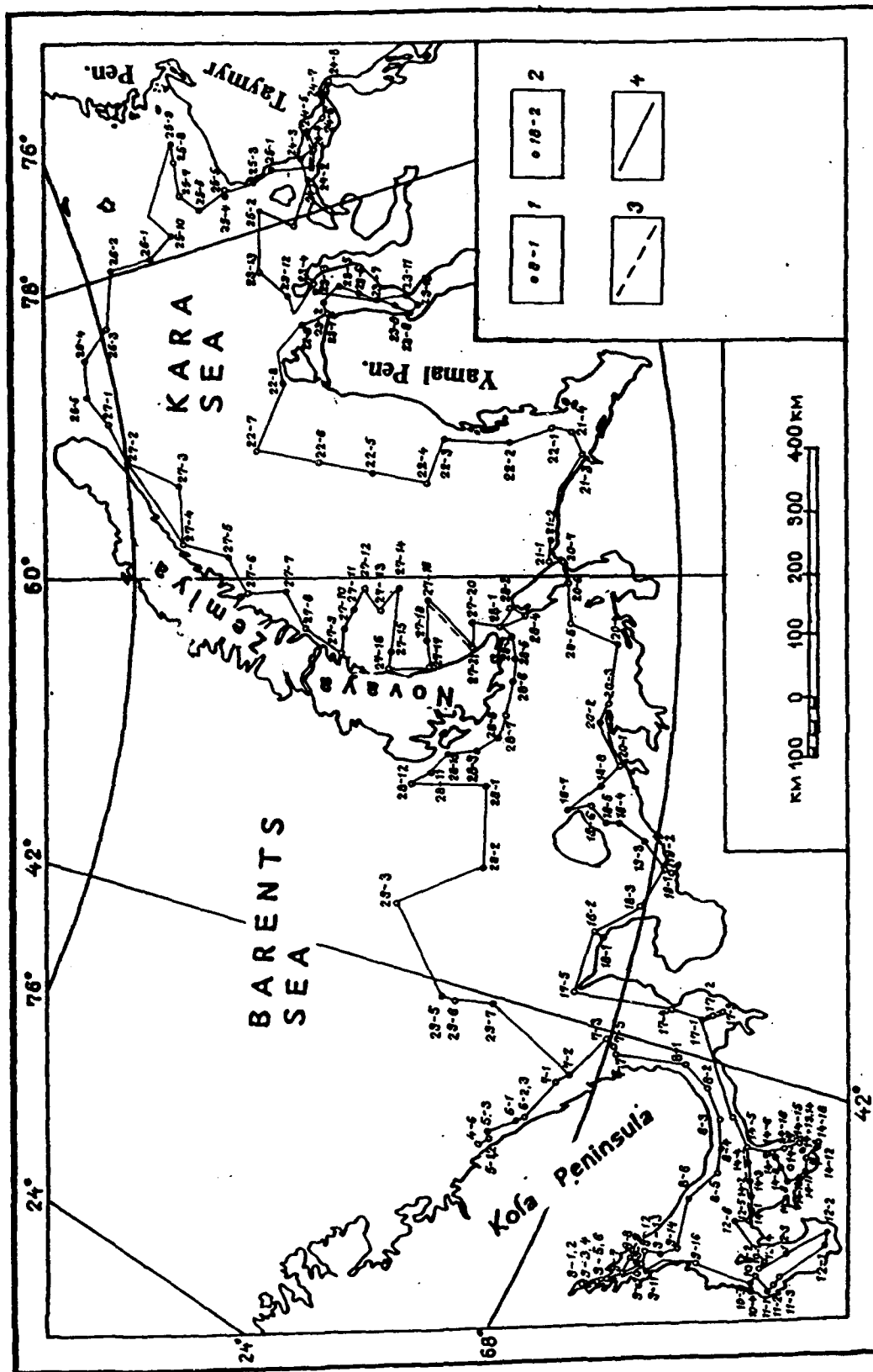


Fig. 1 1991 cruise track and sediment sample locations

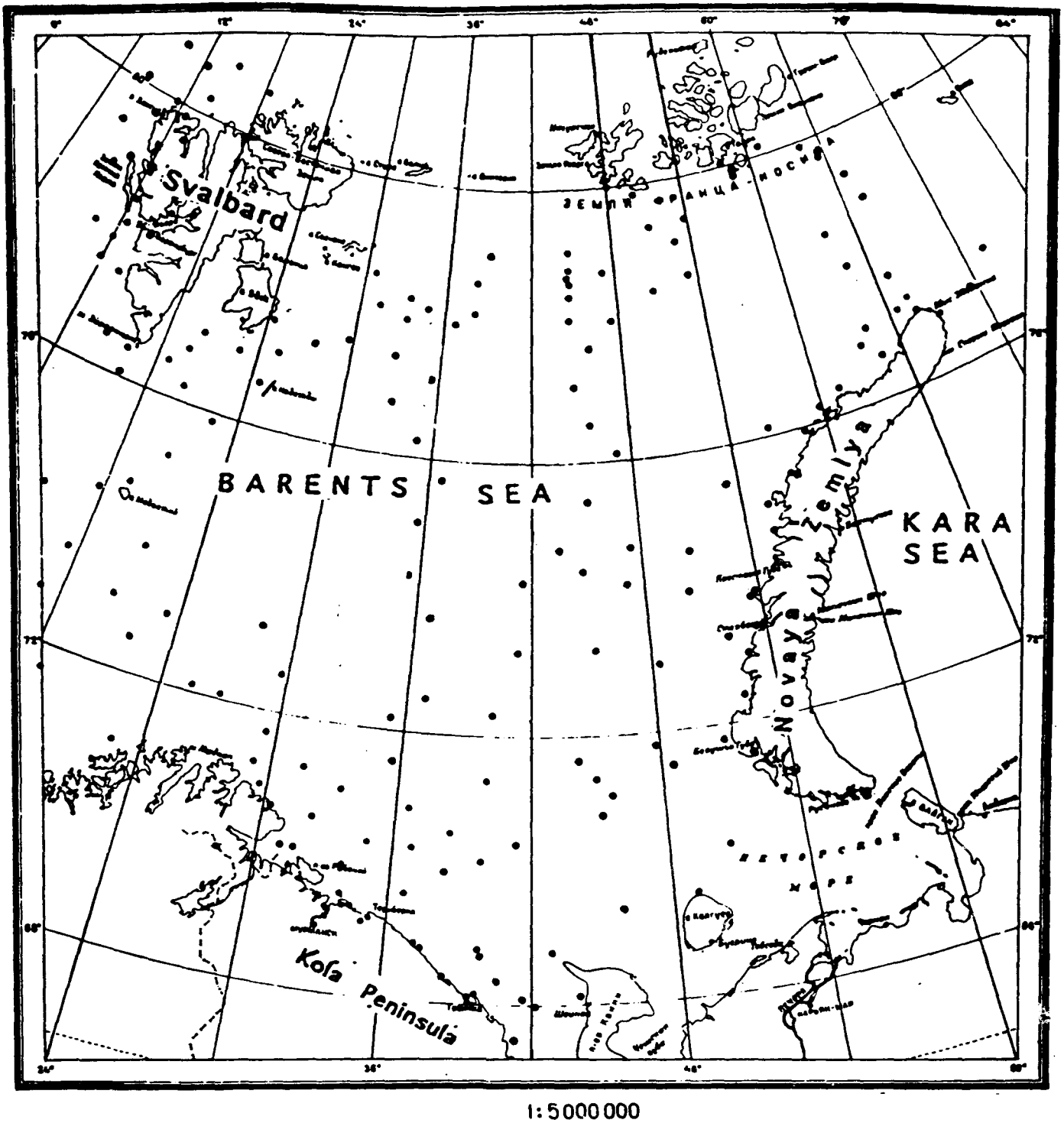


Fig. 2 1992 cruise sediment sample locations

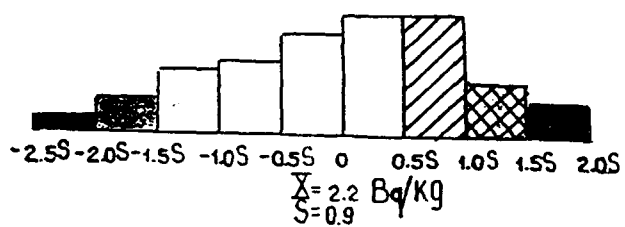
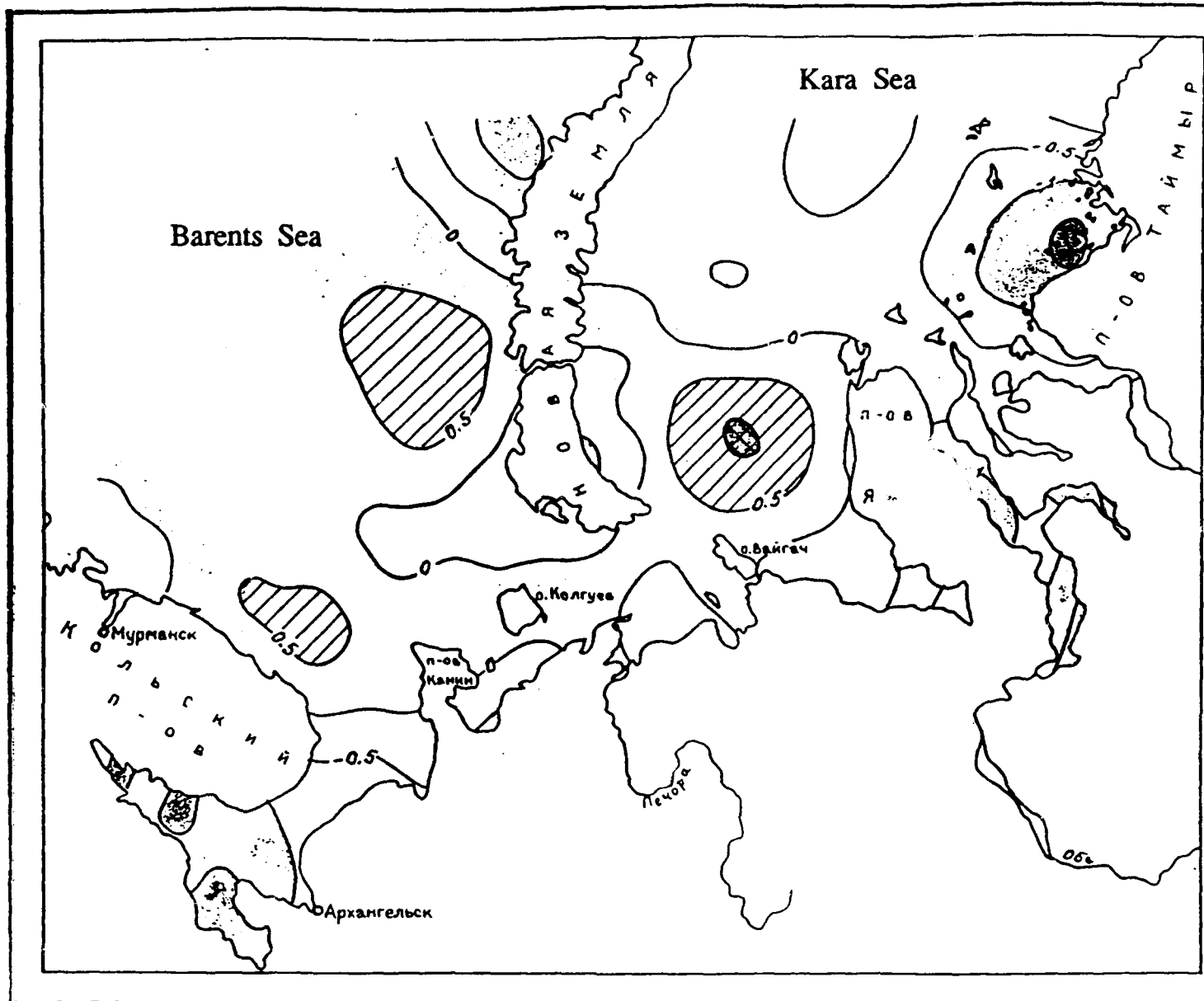


Fig. 3. Distribution of  $^{137}\text{Cs}$  in the Barents and Kara Sea bottom sediments (1991 cruise data) in standard deviation from the mean values (Bq/Kg)

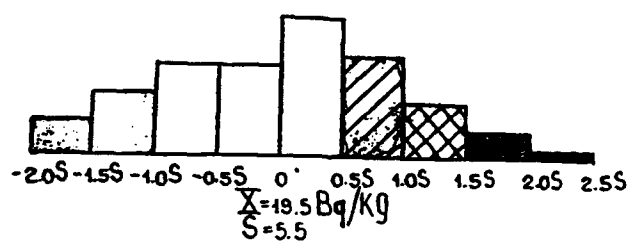
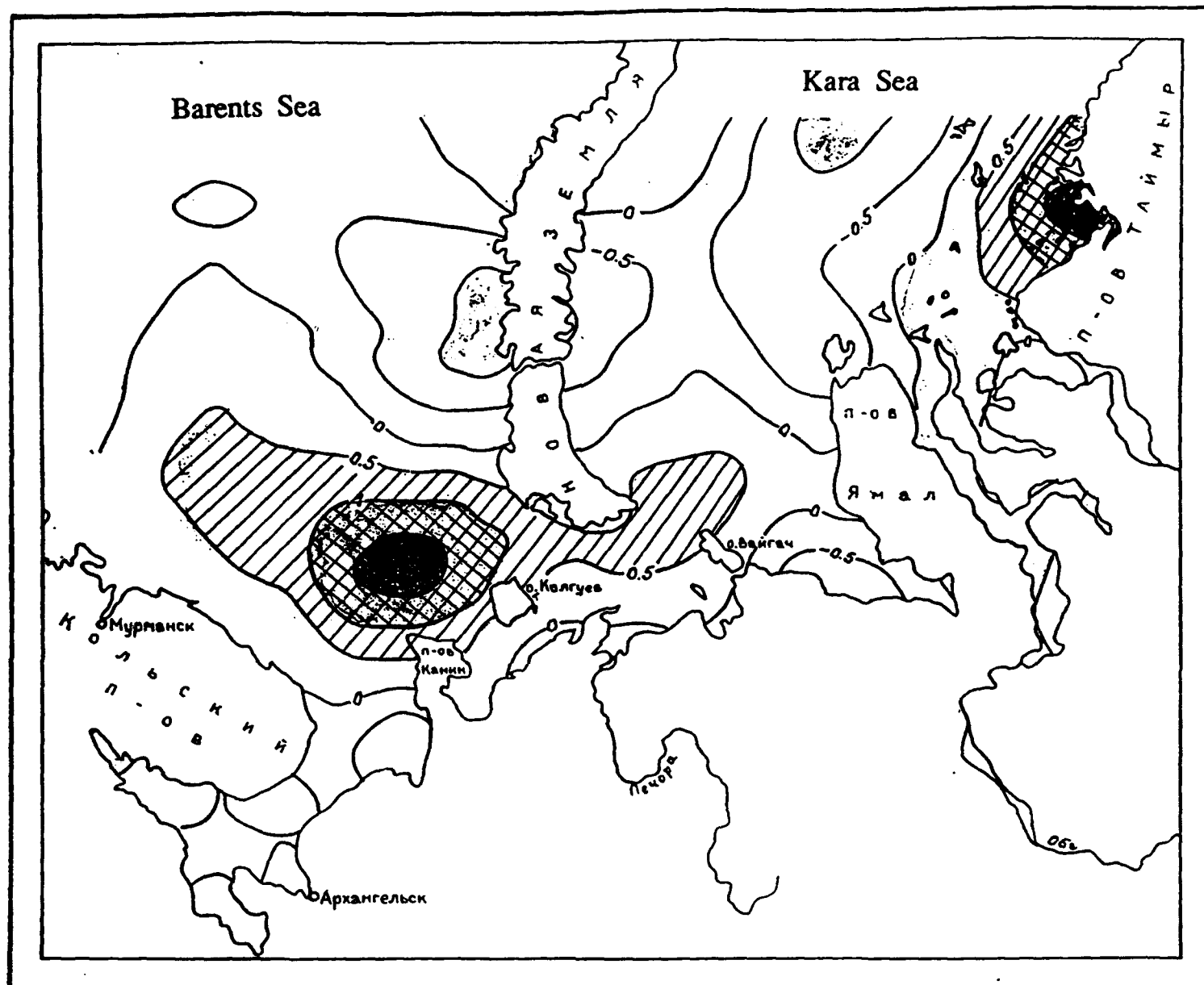


Fig. 4. Distribution of  $^{228}\text{Th}$  in the Barents and Kara Sea bottom sediments (1991 cruise data) in standard deviation from the mean values (Bq/Kg)





### Description of R/V "Geolog Fersman"

Ocean-going double-deck vessel with long forecastle and sternward slip. Diesel powered. Single shaft with controllable pitch propeller in the nozzle.

Length overall	- 103.00 m
Length pp.	- 96.34 m
Beam	- 16.00 m
Draught	- 5.84 m
Displacement	- 5,512 tn
Fuel capacity	- 1,450 tn
Sea endurance	- 100 days
Cruising speed	- 14.50 knots
Maximum speed	- 16.00 knots
Crew	- 53
Scientific personnel	- 39

#### Research equipment:

Main navigation/geophysical system is based on EC-1010 computer and enables continuous ship positioning and real time data processing. Acquisition systems include high resolution seismic, gravity, magnetics, hydroacoustic and hydrological investigations, subwater photography and TV-observations, bottom sampling. Analytical facilities are available for chemical, spectral, X-ray, radiometric and other studies of sediments, minerals and water.

#### Deck equipment:

A variety of geophysical and geological winches, hoisting gear, booms and cranes provide for a wide range of lifting, towing and trawling operations, including coring, dredging and large-volume sampling of bottom sediments.

#### Living conditions:

Single and double cabins with air conditioning; library; gymnasium; well equipped medical facilities.



RADIOACTIVITY AND ENVIRONMENTAL SECURITY IN THE OCEANS:  
NEW RESEARCH AND POLICY PRIORITIES IN THE ARCTIC AND  
NORTH ATLANTIC

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JUNE 7-9, 1993

WOODS HOLE OCEANOGRAPHIC INSTITUTION,  
WOODS HOLE, MA, USA

SELLAFIELD AS A SOURCE OF RADIOACTIVITY  
TO THE BARENTS SEA

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The operation of nuclear facilities in NW Europe has resulted in the controlled release of low-level radioactive waste into coastal waters. The principal sources are due to nuclear fuel reprocessing. This takes place at: La Hague, near Cherbourg, in France discharging into the English Channel; Dounreay in NE Scotland discharging into the Pentland Firth; and, Sellafield (formerly Windscale) on the Cumbrian coast of England, discharging into the Irish Sea. Of these, the Sellafield releases have had the biggest impact and have resulted in a substantial increase in the artificial radionuclide inventory of the North Atlantic (Table 1).

MAFF, as one of the authorising departments, has an extensive monitoring and assessments programme throughout UK coastal waters and beyond. This is backed up by research programmes on the physical, chemical and biological processes controlling the distribution and behaviour of the radionuclides, and on the effects of radiation on aquatic organisms. The results of the monitoring programme are published in an annual report (e.g. MAFF, 1992) which is available on request. The results of the research are published in the open literature.

Discharges began from Sellafield in 1952 and have fluctuated significantly in time, both in terms of the gross radioactivity released and its isotopic composition. Discharges of most radionuclides peaked in the mid- to late-1970s. The caesium peak resulted from a cessation in the reprocessing operation in 1974, requiring prolonged storage of fuel rods under water and consequent corrosion of the cladding and release of greater quantities of the more soluble radionuclides. The introduction of zeolite skips into the storage ponds in 1976 and subsequently a new ion-exchange plant (SIXEP) in 1985 resulted in a very significant decrease in the discharge (Fig. 1).

Regular seawater sampling by MAFF, and the development of compartment models, has allowed the mean  $^{137}\text{Cs}$  concentration in the Irish Sea and North Sea to be estimated (Fig. 2). Not surprisingly they closely match the discharge history.

A large number of studies have been conducted of the radionuclides either directly, to determine their behaviour, or utilising them as tracers of marine processes. Regular German cruises take place to the North Sea, Baltic and more distant waters (Kautsky, 1987; Nies *et al.*, 1991) to define the pattern of Sellafield contamination. Regular French cruises to the Channel and North Sea have established the pattern of contamination from the La Hague plant (Guegueniat *et al.*, 1987) and made it possible to estimate the variable contributions of different sources of seawater to the North Sea from the Channel (La Hague), the Irish Sea (Sellafield), the Atlantic Ocean (fallout) and the Baltic Sea (Chernobyl) on the basis of characteristic isotope signatures (Bailly du Bois *et al.*, 1993). A similar approach may have an application in attempts to establish the influence of various sources of radionuclides in the Arctic region.

Table 1. The Sellafield and La Hague Discharges (TBq):

Nuclide	Sellafield (to 1991) (MAFF data)		La Hague (to 1985) (+)		Fallout in N Atlantic (to 1983) (+)
Cs-137	41000	[29000]	940	[760]	94000
Sr-90	6200	[4200]	755	[675]	64000
Pu-239,240	685	[685]	3 #	[3]	1300
Tc-99	367 *	[367]			

[ ] denotes decay-corrected, environmental inventory

\* includes a pre-1978 estimate of 40 TBq (Aarkrog et al., 1987)

# Pu-alpha

+ Pentreath (1987)

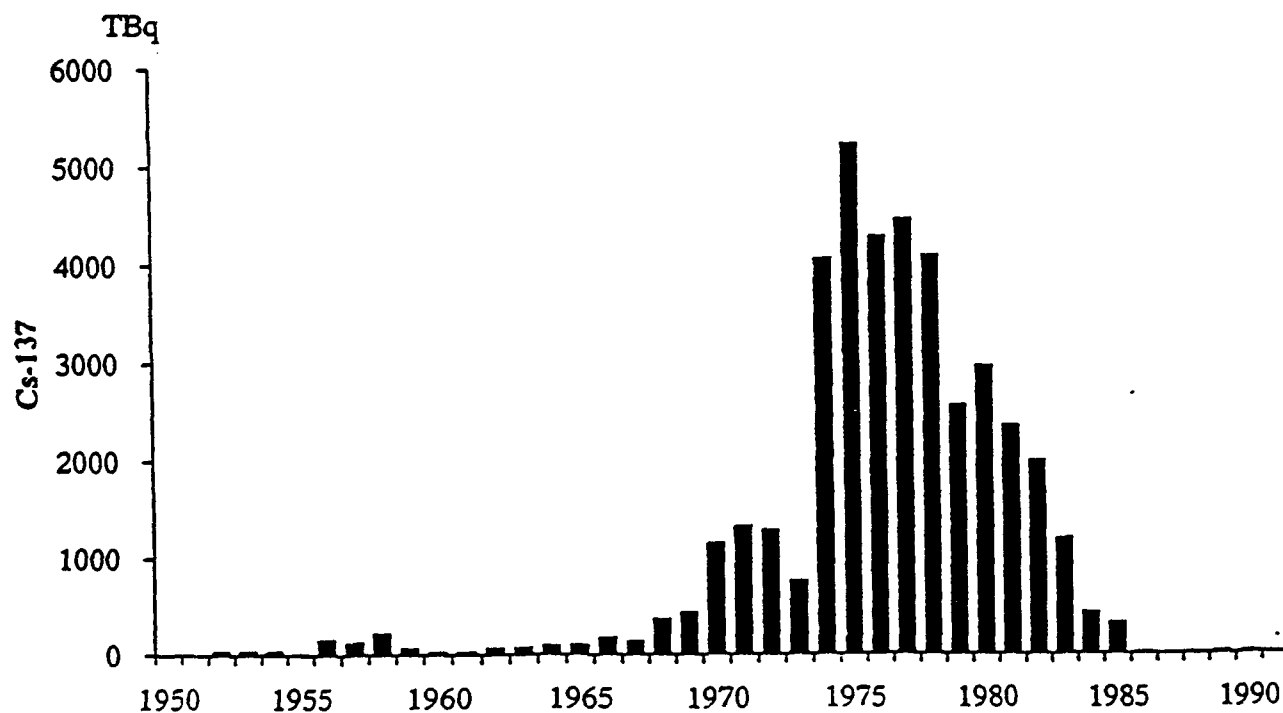


Figure 1. Annual discharges of <sup>137</sup>Cs (TBq) from Sellafield.

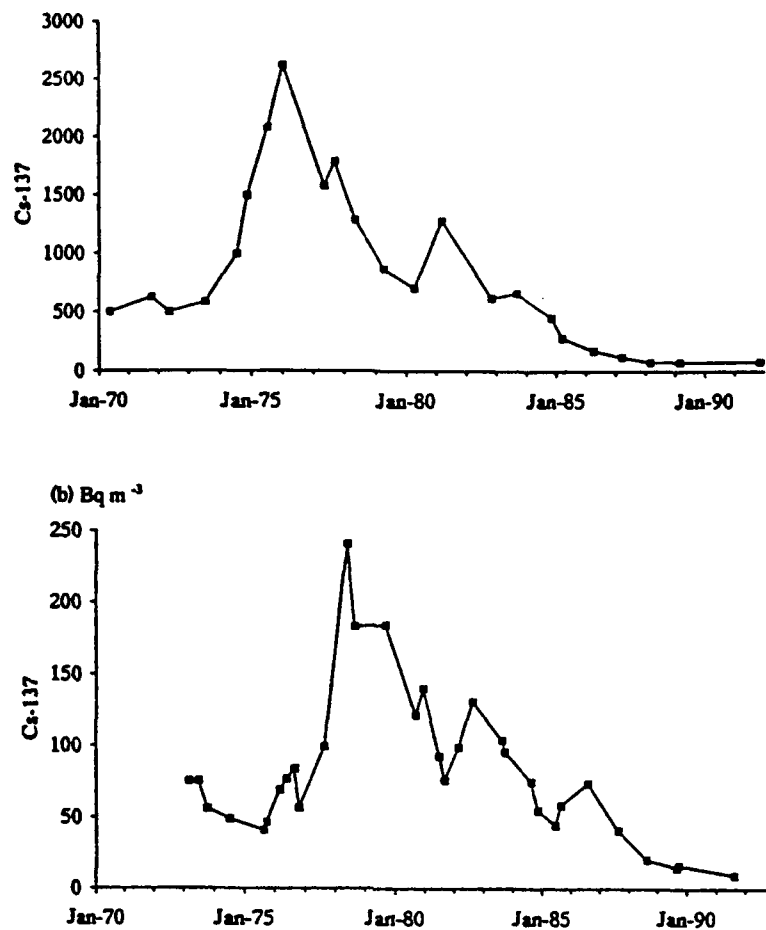


Figure 2. Mean concentrations of  $^{137}\text{Cs}$  (Bq m $^{-3}$ ) in surface waters of: (a) the Irish Sea; and (b) the North Sea.

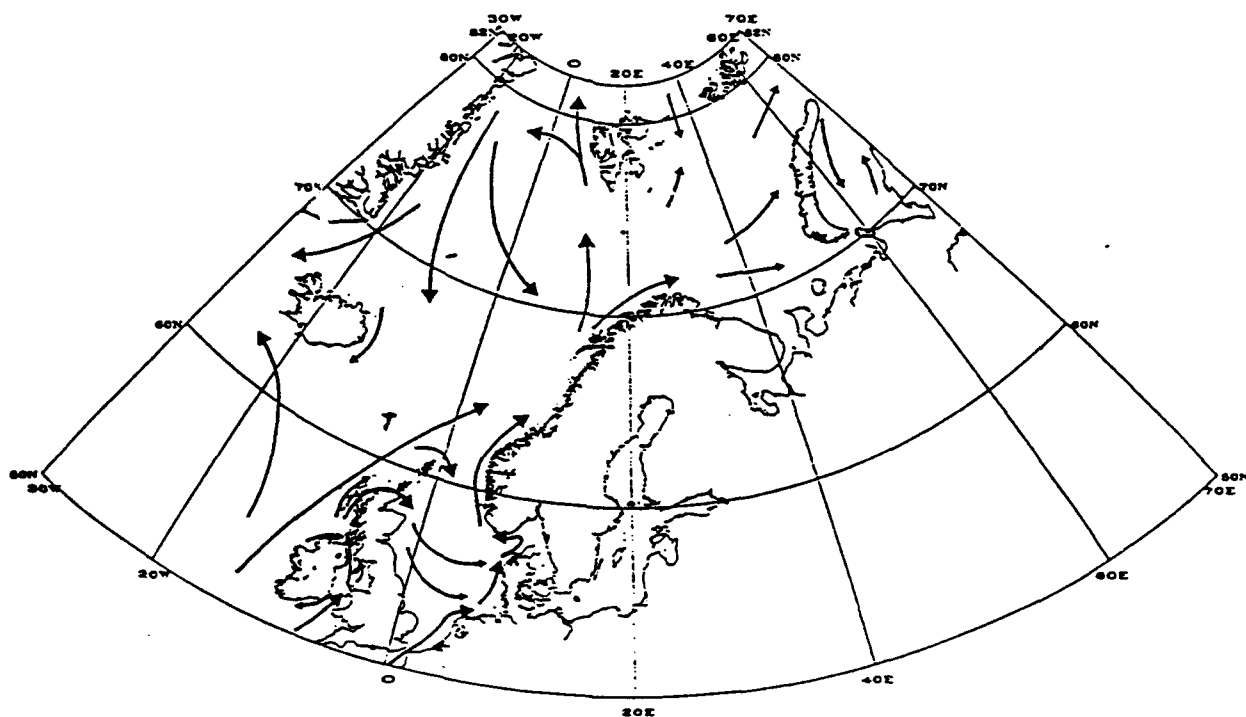


Figure 3. Main surface currents, from various sources.

A summary of the Sellafield discharges, their radiological impact, and most of the relevant investigations which have been conducted of the physical, chemical and biological interactions controlling their behaviour, is contained in a recent review (Kershaw *et al.*, 1992).

## METHODS

MAFF has conducted regular cruises to the Irish Sea and North Sea on virtually an annual basis since the early 1970s. In addition, cruises have taken place to more distant waters at less frequent intervals. Their main purpose has been to obtain data on radiocaesium distributions which, when combined with fish catch data, are used to estimate collective dose commitments from seafood consumption i.e. in a radiological assessment. The value of the data as a tracer of water movement was recognised, however, and later cruises included reliable hydrographic observations. Surface waters were sampled by tapping into the continuously pumped ship's 'clean' seawater supply. Sub-surface samples were obtained with Niskin bottles. More recent cruises have used a CTD-Rosette array.

Most of the data reported here were obtained by pumping 50 litre samples, acidified with nitric acid, through cartridges containing ammonium-duodeca-molydophosphate on silica gel (ASG), and counted using a gated NaI well crystal detector (Baker, 1975). Samples with high levels of radioactivity (e.g. within the Irish Sea) were acidified with hydrochloric acid and passed through cartridges containing potassium cobaltihexacyanoferrate (KCFC) to avoid interference from  $^{95}\text{Zr}/^{95}\text{Nb}$ . The method has been shown to perform satisfactorily in intercomparison exercises (Steele, 1989).

## RADIOCAESIUM DISTRIBUTIONS AND TRANSPORT PATHWAYS

Radionuclides have differing affinities for adsorption onto particle surfaces. A large fraction of both short- (e.g.  $^{95}\text{Zr}/^{95}\text{Nb}$ ) and long-lived radionuclides (e.g.  $^{239,240}\text{Pu}$ ,  $^{241}\text{Am}$ ) are readily incorporated into the seabed sediments of the Irish Sea. However the more soluble radionuclides, such as radiocaesium and  $^{99}\text{Tc}$ , are transported considerable distances from the Irish Sea: via Scottish waters to the North Sea, Norwegian Sea, Barents Sea, Arctic Ocean, East Greenland Current and Baffin Bay (e.g. Livingston *et al.*, 1982, 1984; McKay *et al.*, 1986; Holm *et al.*, 1986; Hallstadius *et al.*, 1986; Dahlggaard *et al.*, 1986, 1991; Aarkrog *et al.*, 1987; Kautsky, 1987). A schematic representation of the main surface water currents is shown in Figure 3.

Such studies have demonstrated the contribution of Sellafield-derived radionuclides to the inventory of artificial radionuclides at high latitudes and provided a means of estimating both dilution factors and transit times (see Table 2). Radiocaesium has been widely used in modelling studies of the water circulation over much of the NW European Shelf (e.g. Prandle & Beechey, 1991).

MAFF conducted 5 cruises to the Barents Sea in the period 1975-1989, in the months of August to September. These data have been combined with data from separate, but more or less contemporaneous, MAFF cruises to the North Sea and Scottish waters to give a clearer pattern of the development of the Sellafield radiocaesium signal from UK waters to the Barents Sea (Fig. 4). The MAFF data have been published in a series of reports (Camplin & Steele, 1991; Baxter *et al.*, 1992; Baxter & Camplin, 1993). In addition, determinations of  $^{238}\text{Pu}$ ,  $^{239,240}\text{Pu}$  and  $^{241}\text{Am}$  were carried out on a limited number of filtrate and particulate samples from the Barents Sea in 1981, 1985 and 1989. These data, together with radiocaesium data from the Greenland Sea, will be reported separately.

The passage of the 1970's radiocaesium discharge peak is clearly visible. But, we do not have data from the Barents Sea which obviously pre-dates the arrival of the pulse. The Sellafield signal is quite closely confined to the Scottish coastal current, entering the North Sea principally via the Pentland Firth. Various transport routes are conceivable across the North Sea. Residence times in the central North Sea can be significantly longer than in the waters immediately to the north and south. There is a quite distinct boundary between North Sea water and water entering via the English Channel. Once in the Norwegian coastal current radiocaesium appears to be transported rapidly north, with considerable lateral mixing (due to the development of eddies), resulting in a broadening of the region contaminated by the Sellafield signal. The current splits around North Cape: one branch heading north to the west of Svalbard; the other following the coast eastwards. This is reflected in the  $^{137}\text{Cs}$  distribution. MAFF have not been able to sample east of  $30^\circ\text{E}$ .

The 1989 cruise included  $\gamma$ -probe profiles and sampling around the site of the sunken Komsomolets submarine, 7 months after the accident. At that time, with the methods available, it was not possible to detect any leakage of radiocaesium from the vessel (Camplin & Read, 1992).

The  $^{137}\text{Cs}$  vs salinity relationship, shown in Figure 5 for 1989, shows the mixing and dilution of relatively low salinity, high  $^{137}\text{Cs}$ -contaminated coastal waters with Atlantic water. The cluster of points at a salinity of 34.92 represents Norwegian Sea Deep Water with uniformly low  $^{137}\text{Cs}$  concentrations.

## TIME-SERIES

Long time-series are invaluable when seeking a better understanding of the rates and continuity of processes. The studies by Aarkrog and co-workers of  $^{99}\text{Tc}$  in fucoid seaweeds are particularly notable (e.g. Aarkrog *et al.*, 1987).

In this study we present MAFF data from 5 regions: the North Channel (monthly sampling from 1972-1986, quarterly and/or RV cruises at other times); East Scotland, in box defined as  $57.0\text{-}57.5^\circ\text{N}$ ,  $1.5\text{-}2.0^\circ\text{W}$  (monthly sampling from 1982-1992 plus RV cruises); a central North Sea box, defined as  $56^\circ\text{-}58^\circ\text{N}$   $0^\circ\text{-}4^\circ\text{E}$  (RV cruises); West Norway, in a box



Figure 4. Distribution of  $^{137}\text{Cs}$  ( $\text{Bq m}^{-3}$ ) in surface waters: 1975, 1979, 1981, 1985 and 1989.

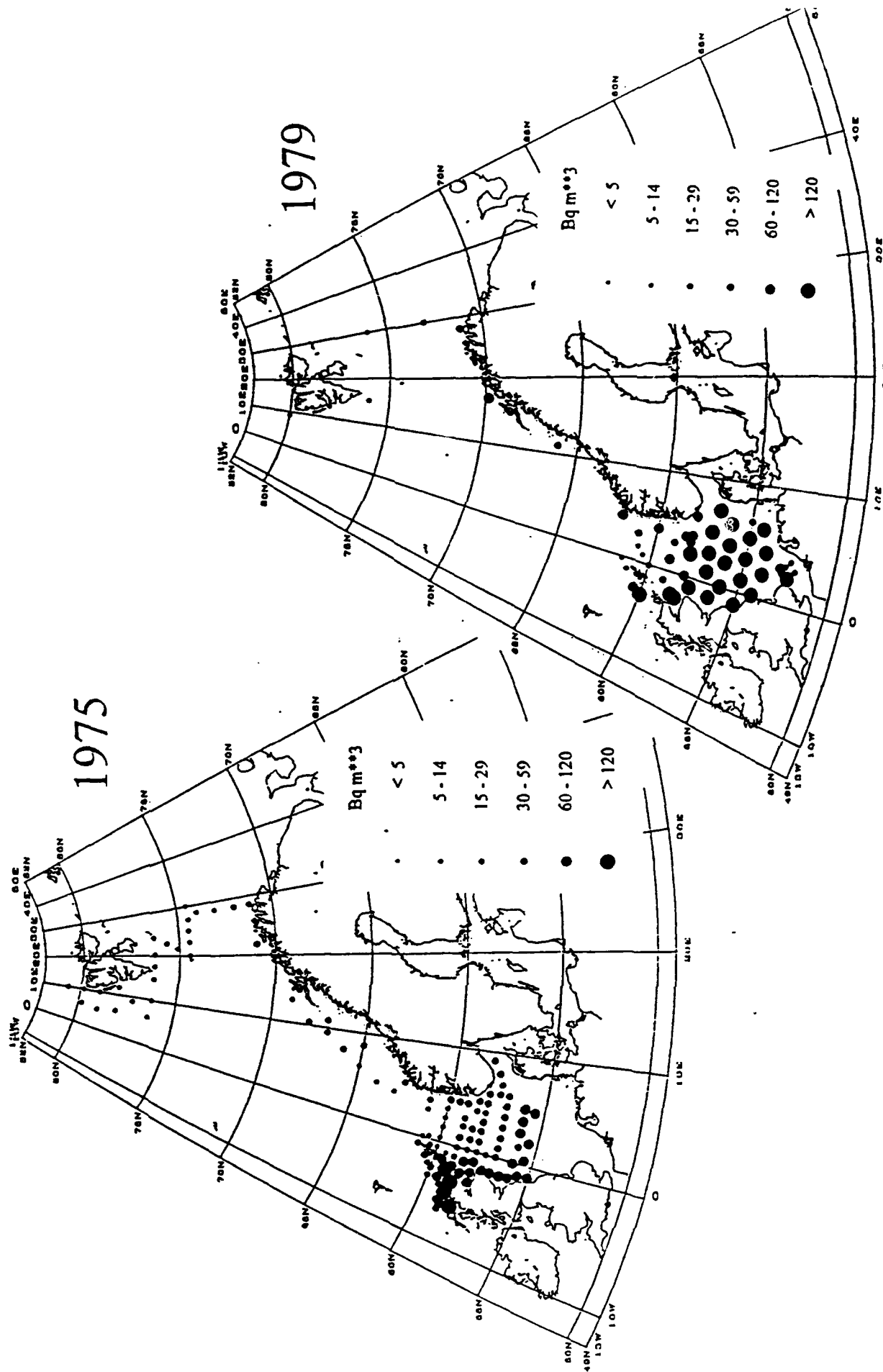
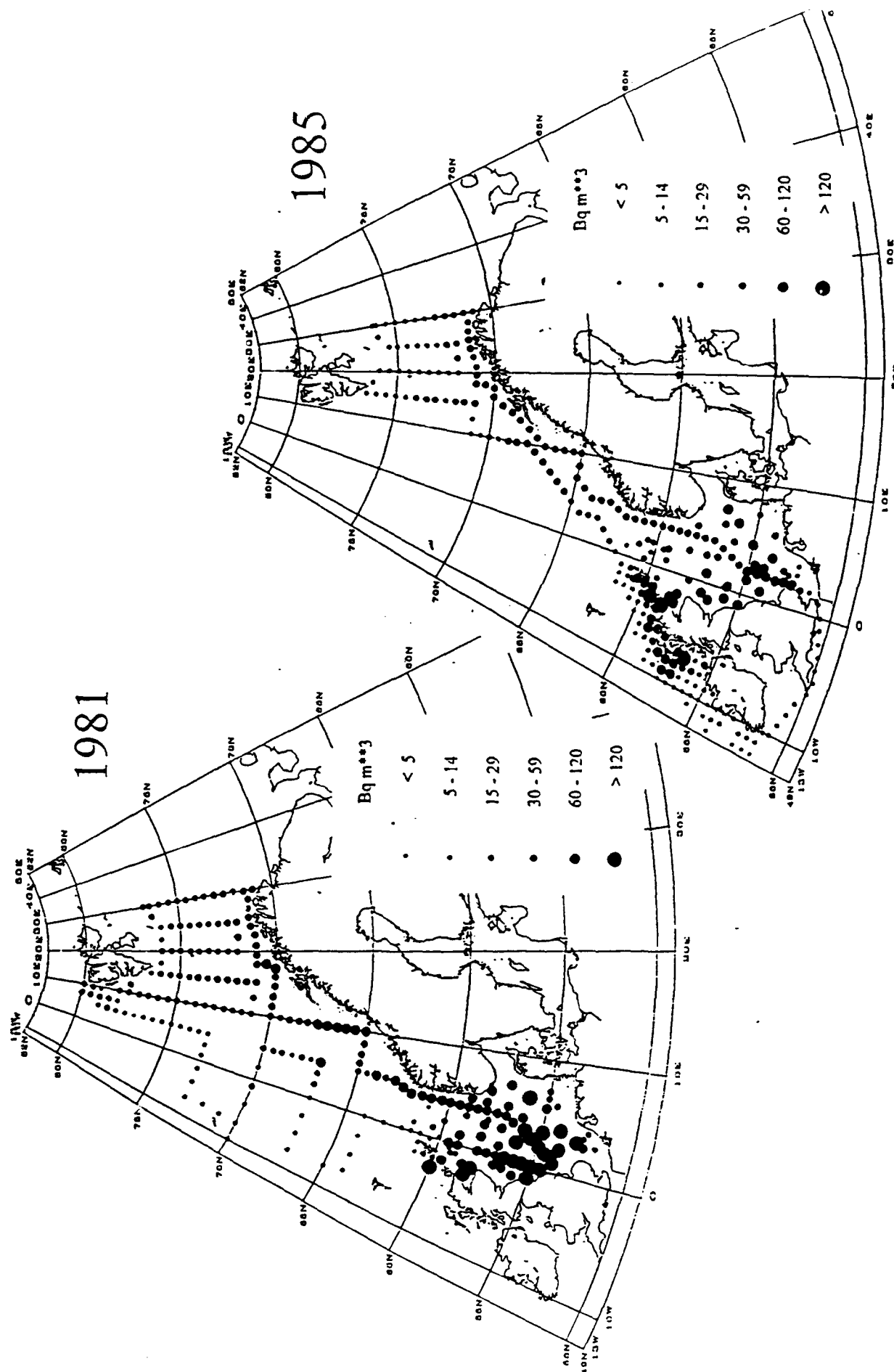


Figure 4. Continued.



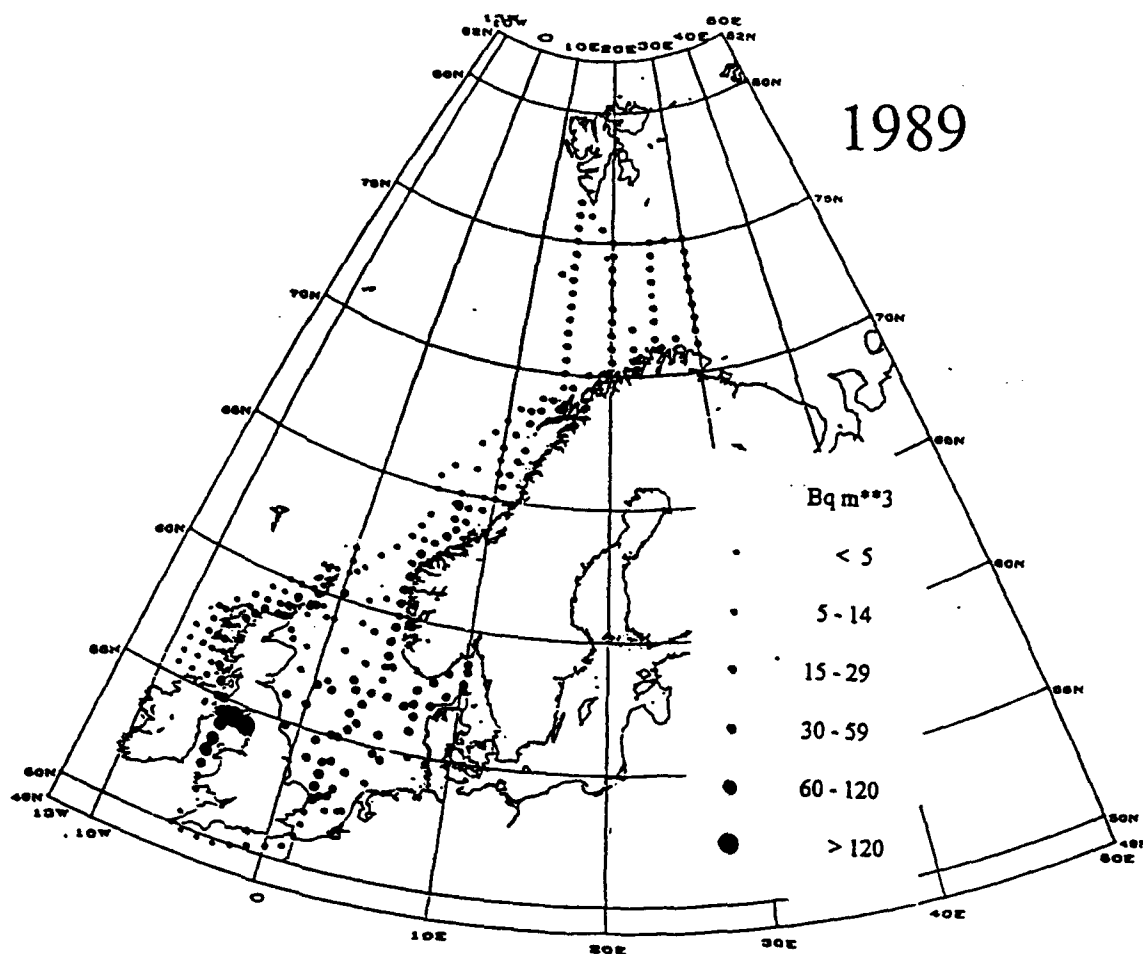
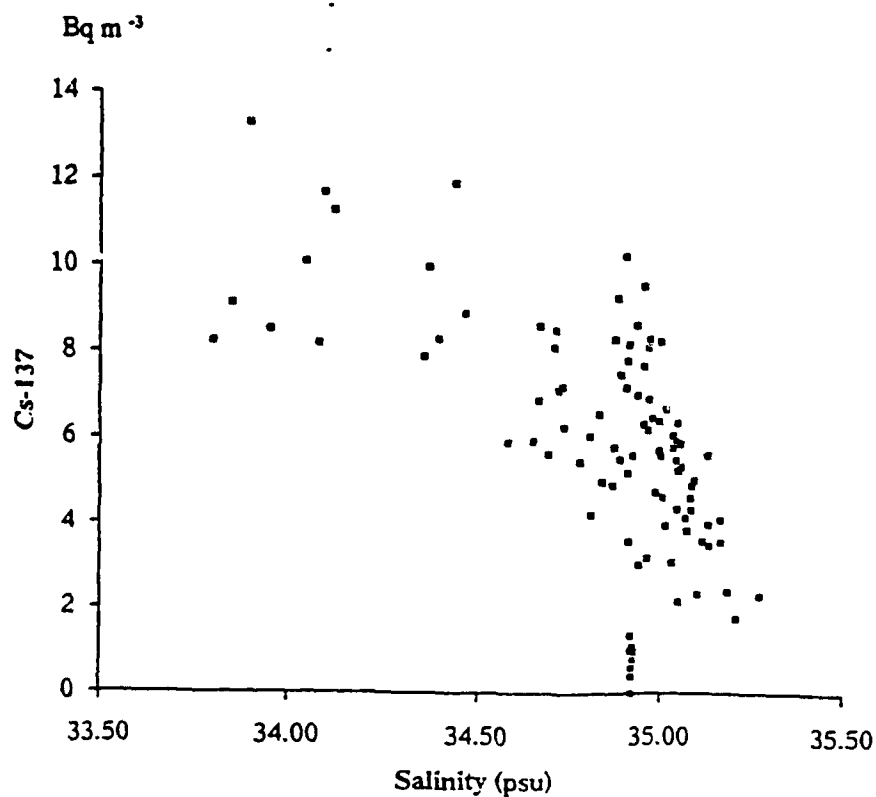
Figure 5.  $^{137}\text{Cs}$  ( $\text{Bq m}^{-3}$ ) vs salinity (psu) relationship in the Barents Sea and Norwegian Sea, 1989.

Figure 6. The time-dependent variation of  $^{137}\text{Cs}$  concentrations ( $\text{Bq m}^{-3}$ ) in surface waters, with the Sellafield discharge (TBq) shown for reference.

100

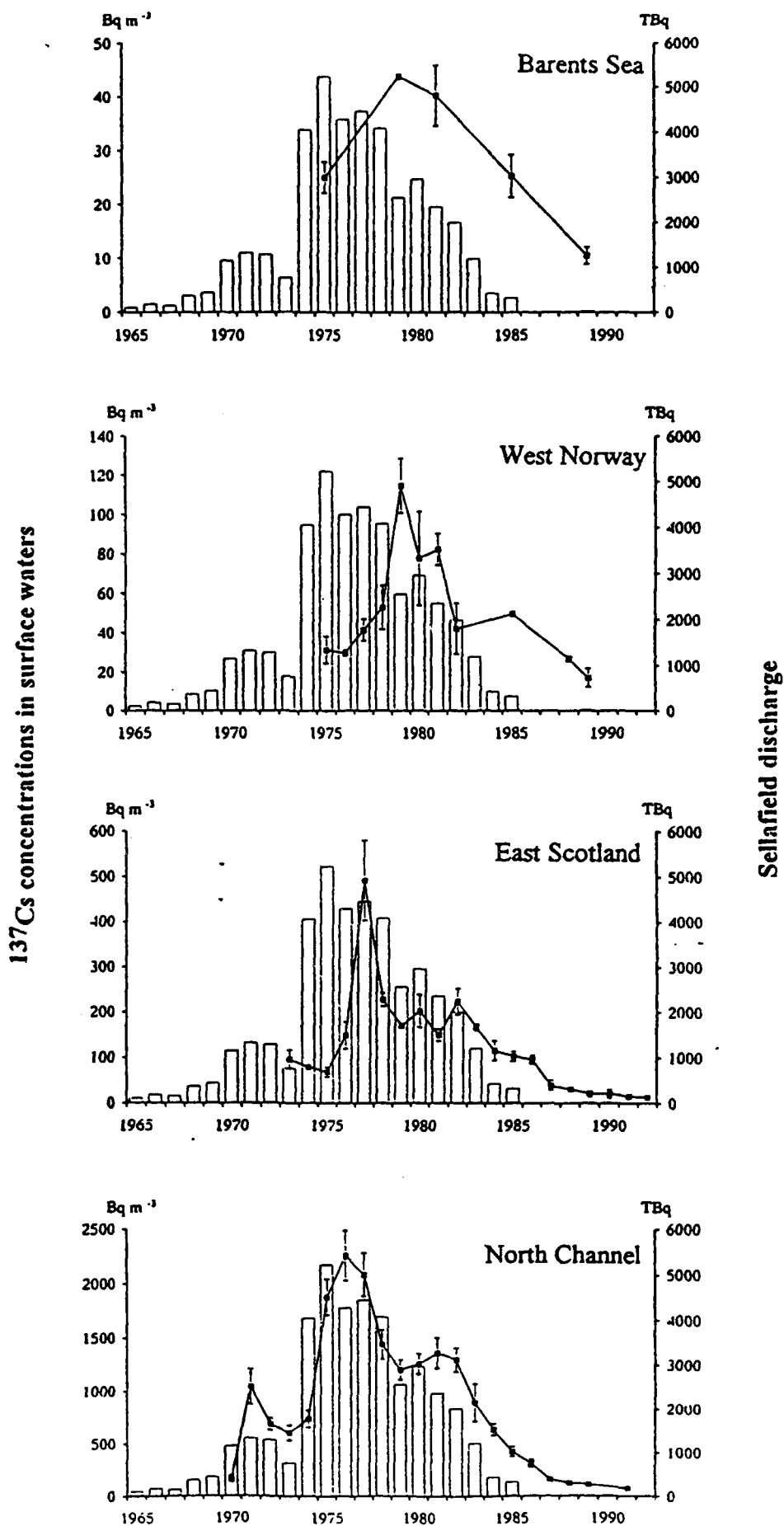


Table 2. Approximate Transit Times from Sellafield to:

North Channel	1y	Cs-137	This study
East Scotland	2-3y	"	"
West Norway	3-4y	"	"
West Barents Sea	4-5y	"	"
Barents Sea	5y	Cs-137	Livingston et al. (1984)
Svalbard	5y	"	Dahlgaard et al. (1986)
East Greenland	7y	Cs-134, Tc-99	Dahlgaard et al. (1986) and Aarkrog et al. (1987)
Baffin Bay	8y	" "	Aarkrog et al. (1987)
Arctic Ocean, 1500m depth, LOREX	8y	Cs-137	Livingston et al. (1984)

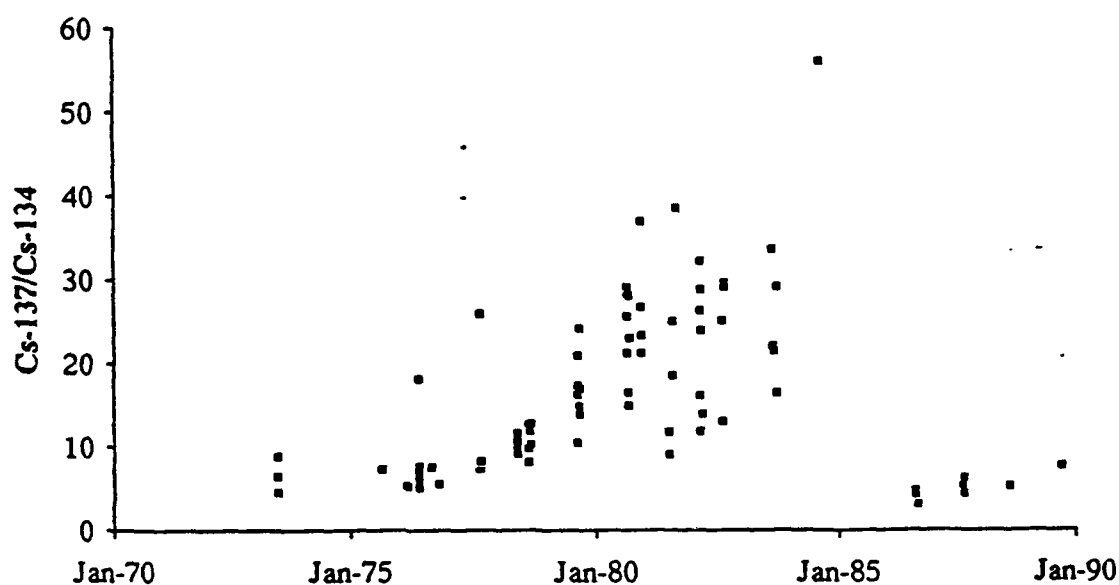


Figure 7. Variation of the  $^{137}\text{Cs}:^{134}\text{Cs}$  ratio with time in the central North Sea (56°-58°N, 0°-4°E).

defined as 59°-61°N, 3.5°-5°E (RV cruises); and, SW Barents Sea, in a box defined as 71°-72°N, 20°-30°E (RV cruises).

Figure 6 shows the variation of the mean  $^{137}\text{Cs}$  concentration in surface waters with time in the North Channel, East Scotland, West Norway and Barents Sea boxes. The error bars represent  $\pm 2 \sigma$  standard error. The pattern of the Sellafield discharge is clearly visible in the boxes closer to the source. The pattern is still evident in the Barents Sea but rather poorly defined because of the paucity of time-series data. On the basis of these figures, crude estimates have been made of transit times. These are presented in Table 2, together with transit times from previous studies in high latitudes. Those estimated by Kautsky (e.g. Kautsky, 1987) have been omitted on the grounds that, although internally consistent, they tend to be significantly longer than other studies would suggest. There is broad agreement between the transit times deduced from the MAFF data and earlier work in Scottish waters and the North Sea (e.g. Dahlgaard *et al.*, 1986; Hallstadius *et al.*, 1986).

The impact of Chernobyl on the waters of the NW European Shelf has been reported (e.g. Nies and Wedekind, 1987; Camplin *et al.*, 1986). Figure 7 shows the perturbation to the  $^{137}\text{Cs}/^{134}\text{Cs}$  ratio caused by an influx of Chernobyl radiocaesium to the central North Sea, interrupting the upwards trend of the ratio due to the Sellafield signal. The scatter in the data is partly due to the variable influx of Atlantic water across the northern boundary of the defined box.  $^{134}\text{Cs}$  data are rather lacking at higher latitudes in the MAFF dataset (because of the low concentrations and relatively small sample size) so it has not been possible to readily account for the influence of the Chernobyl input on the 1989 cruise data.

## CONCLUSION

MAFF data on radiocaesium distributions across the NW European Shelf support the conclusions of previous studies that the low-level radioactive waste discharges from Sellafield, into the Irish Sea, are transported to the Barents Sea and beyond. A transit time of 4-5 years is proposed from Sellafield to the Western Barents Sea (specially a box defined as 71°-72°N, 20°-30°E). This data set should be combined with others, particularly for the period of the 1960s and 1970s, to allow a more comprehensive assessment of the relative importance of the Sellafield discharge as a source of radioactivity to this region in the past, and the likely extent of its influence in the future.

## ACKNOWLEDGEMENTS

The authors wish to record their appreciation of the efforts and dedication of the many members of the scientific staff and ship's crew who assisted in the collection of samples and/or production of the radiocaesium and hydrographic data. The contributions of Doug Jefferies, John Dutton, Ken Steele and Carroll Baker, all former colleagues at the Fisheries Laboratory, is warmly acknowledged. Bob Dickson and Den Woodhead provided helpful advice and

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**Estimated Inventory of Radionuclides in  
Former Soviet Union Naval Reactors  
Dumped in the Kara Sea  
and Their Associated Health Risk**

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**Abstract**

Radionuclide inventories have been estimated for the reactor cores, reactor components, and primary system corrosion products in the former Soviet Union naval reactors dumped at the Abrosimov Inlet, Tsivolka Inlet, Stepovoy Inlet, Techeniye Inlet, and Novaya Zemlya Depression sites in the Kara Sea between 1965 and 1988. For the time of disposal, the inventories are estimated at 17 to 66 kCi of actinides plus daughters and 1,695 to 4,782 kCi of fission products in the reactor cores, 917 to 1,127 kCi of activation products in the reactor components, and 1.4 to 1.6 kCi of activation products in the primary system corrosion products. At the present time, the inventories are estimated to have decreased to 6 to 24 kCi of actinides plus daughters and 492 to 540 kCi of fission products in the reactor cores, 124 to 126 kCi of activation products in the reactor components, and 0.16 to 0.17 kCi of activation products in the primary system corrosion products. All actinide activities are estimated to be within a factor of two.

We have also conducted a preliminary risk assessment of key actinides and fission products in the discarded spent nuclear fuel as a means of identifying which radionuclides are most important from a human-health standpoint. Results of such an assessment can also be used to guide future monitoring programs conducted in Arctic waters. Global population doses resulting from the release of radionuclides contained in the reactors were estimated using simple dose-conversion factors (developed originally by UNSCEAR) that provide estimates of collective dose commitments for unit releases of radionuclides to sea water. The estimated population doses using the appropriate dose conversion factors and the estimated inventories are 2.3 person-Sv for  $^{90}\text{Sr}$ , 4.2 person-Sv for  $^{241}\text{Am}$ , 5.2 person-Sv for  $^{137}\text{Cs}$ , and 0.1 person-Sv for  $^{239}\text{Pu}$ . One interesting result is that although the inventory of  $^{241}\text{Am}$  is much lower than the inventory of  $^{90}\text{Sr}$ ,  $^{241}\text{Am}$  has a greater predicted collective dose commitment because of a higher dose-conversion factor. Finally, based on a cancer-risk factor of 0.05/Sv, we calculate a global risk of 0.6 fatal cancers for release of the key actinides and fission products. By comparison, the population risk for the Chernobyl accident has been estimated to be 17,000 fatal cancers.

In the Spring of 1993, a Russian report, "Facts and Problems Related to Radioactive Waste Disposal in Seas Adjacent to the Territory of the Russian Federation,"<sup>1</sup> was released. The findings presented in this Russian report were the result of a scientific study commissioned by the Office of the President of the Russian Federation and headed by Dr. Alexi V. Yablokov. The Yablokov Commission, as they were later called, reported that 16 naval reactors from seven former Soviet Union submarines and the icebreaker *Lenin*, each of which suffered some form of reactor accident, were dumped at five sites in the Kara Sea. Six of these 16 naval reactors contained their spent nuclear fuel (SNF). In addition, approximately 60% of the SNF from one of the three *Lenin* naval reactors was disposed of in a reinforced concrete container and metal shell. The Yablokov Commission estimates of radioactivity were limited to the fission products in the SNF and the <sup>60</sup>Co in the reactor components, both at the time of disposal. With rare exception, specific radionuclides were not identified and there was no estimate provided for the current levels of radioactivity.

This report presents the results of an independent effort to provide a time-dependent inventory of the actinides, fission products, and activation products in these 16 former Soviet Union naval reactors and their SNF, and a preliminary risk assessment of the health risks to man from the release of key actinides and fission products.

### Background Information

The information presented herein highlights the conclusions of the Yablokov Commission and what we know or have assumed about the history of each submarine. Table 1 presents the Yablokov Commission findings for the five Kara Sea disposal sites.<sup>1</sup> Summarized for each disposal site is the disposal date, the number of discarded naval reactors and their associated ship identification number, the number of discarded naval reactors containing SNF, and the estimated fission product radioactivity in the SNF at the time of disposal. The Tsivolka Inlet entries are for the icebreaker *Lenin* and the reinforced concrete container and metal shell containing approximately 60% of the SNF from one of the three OK-150 power plant naval reactors that were discarded in 1967. The 100 kCi reported for the *Lenin* disposal result primarily from the fission products <sup>90</sup>Sr and <sup>137</sup>Cs. The two naval reactors containing SNF that were discarded in the Stepovoy Inlet in 1981 are identified as being of a liquid metal cooled type. The Yablokov Commission estimates of total radioactivity are 2,300 kCi of fission products in the SNF and 100 kCi of <sup>60</sup>Co in the reactor components. No information was provided which would allow association of a given ship identification number with a specific submarine class or accident date.

**Table 1. Yablokov Commission findings for the former Soviet Union naval reactors dumped in the Kara Sea.<sup>1</sup>**

Disposal Site	Disposal Date	Naval Reactors Discarded	Reactors Containing SNF	Fission Product Activity (kCi)
Abrosimov Inlet	1965	2 (No. 285)	1	800
		2 (No. 901)	2	400
		2 (No. 254)	-	-
	1966	2 (No. 260)	-	-
Tsivolka Inlet	1967	3 (OK-150)	0.6*	100
Novaya Zemlya Depression	1972	1 (No. 421)	1	800
Stepovoy Inlet	1981	2 (No. 601)	2	200
Techeniye Inlet	1988	2 (No. 538)	-	-
<b>Total</b>	<b>16</b>		<b>6.6</b>	<b>2,300</b>

\*The SNF was not contained in the naval reactor, but in a reinforced concrete and metal shell.

To estimate the time-dependent inventory of radionuclides in the discarded naval reactors, reactor core operating histories and the accident date associated with each discarded naval reactor are required. Unfortunately, reactor core histories for the seven former Soviet Union submarines were not available. Therefore, an analytical model was developed to estimate the minimum reactor fuel load for each submarine whose discarded naval reactors contained SNF. As will be discussed later, the model uses as its basis Western estimates of the shaft horsepower of each submarine involved. Selection of an appropriate shaft horsepower requires a knowledge of each submarine's NATO classification.

Table 2 presents a summary of the Western estimates of the identities of the submarines whose naval reactors were dumped in the Kara Sea.<sup>2,3</sup> Summarized for each submarine is the K identification number, NATO classification, and associated reactor accident date. The two naval reactors in the K-27 are reported to have been of a liquid metal type.<sup>2</sup> All other discarded naval reactors are believed to have been of the pressurized water reactor (PWR) type.<sup>4</sup> Three of these submarines, K-3, K-11, and K-19, were observed in active service some years after suffering their reactor accidents. While each of the seven identified submarines was reported to have suffered some form of reactor accident, none was reported to have sunk.

**Table 2. Western estimates of the identities of the former Soviet Union submarines whose naval reactors were dumped in the Kara Sea.<sup>2,3</sup>**

Submarine Identification	NATO Classification	Reactor Accident Date
K-3	November	June, 1962 September 8, 1967
K-5	Hotel/November	Mid-1960s
K-11	November	February 12, 1965
K-19	Hotel	July 4, 1961
K-22*	—	—
K-27	November	May 24, 1968
K-140	Yankee II	August 23, 1968

\*No information is currently available in the open literature for this submarine.

With the information of Table 2 as a basis, a NATO classification was assigned to the ship identification of each submarine whose discarded naval reactors contained SNF. Table 3 presents a summary of our deductions. Summarized for each disposal date is the number of discarded naval reactors containing SNF and associated ship identification number, the K identification number, and the NATO classification. The rationale for our selections is as follows. A recent International Atomic Energy Agency publication<sup>3</sup> identifies three of the four submarines whose naval reactors were discarded in 1965 and 1966 as the K-3, K-11, and K-19. In addition, the submarine whose two naval reactors were discarded in 1981 is identified as the K-27. Since the Yablokov Commission report specified that the minimum time period between reactor shutdown and disposal was one year, we believe that the two submarines associated with the three naval reactors containing SNF that were discarded in 1965 are the K-3 and K-19. Since the first K-3 submarine reactor accident involved no fatalities and she was observed in active service some years later,<sup>2</sup> one may infer that while both naval reactors were undoubtedly replaced, only one of the two discarded naval reactors contained SNF. Furthermore, since the K-19 submarine reactor accident involved fatalities, the accident was of such severity that she was nicknamed "Hiroshima," and she was observed in active service some years later,<sup>2</sup> one may infer that both naval reactors were removed and that each contained SNF. Thus, the K-3 was assigned to the ship identified as No. 285, and the K-19 was assigned to the ship identified as No. 901. Through a similar process of elimination, the submarine associated with the one naval reactor containing SNF that was discarded in 1972 was assigned to the K-140. The three remaining submarines, K-5, K-11, and K-22, are assumed to be associated with discarded naval reactors without SNF.

**Table 3. Best estimate association of ship identification with the NATO classification of each submarine whose discarded naval reactors contained SNF.**

Disposal Date	Reactors Containing SNF	Submarine Identification	NATO Classification
1965	1 (No. 285)	K-3	November
	2 (No. 901)	K-19	Hotel
1972	1 (No. 421)	K-140	Yankee II
1981	2 (No. 601)	K-27	November

#### Analytical Model

The information presented herein describes (1) the analytical model used to estimate the minimum reactor fuel load for each submarine whose discarded naval reactors contained SNF, (2) the information that we know or have assumed about the operating characteristics of the icebreaker *Lenin* and each submarine whose discarded naval reactors contained SNF, and (3) the method used to predict the activation product inventories in the reactor components and primary system corrosion products of all discarded naval reactors.

With an estimate of the reactor fuel load, the reactor power, and the reactor operating history, one can proceed to calculate the radionuclide inventory associated with the SNF. Before describing the computer code that was used to estimate the inventory, the information that is required as input must be addressed. In the case of the icebreaker *Lenin*, core history information necessary to the inventory calculations was directly available from Russian sources.<sup>5,6</sup> Table 4 presents a summary of the naval reactor core information for the icebreaker *Lenin*. Summarized for each of the three *Lenin* reactors is the <sup>235</sup>U loading, the operating period, and the number of effective full power hours. From the information contained in Table 4, the average full power of each reactor is calculated to be 65 megawatts thermal (MW). Each of the three *Lenin* reactors were reported to contain 219 fuel assemblies with a <sup>235</sup>U enrichment in the range of 4.6 to 6.4%. The reactor accident that precipitated the need for disposal of the three naval reactors and a portion of one's SNF occurred either early or late in the year of 1966, some three years after the reactors were refueled. The Yablokov Commission report states that SNF from 125 fuel assemblies, or approximately 60% of the fuel complement from one OK-150 reactor, was discarded. The number of fuel assemblies that this 60% finding implies is on the order of 208, which is in excellent agreement with the 219 fuel assemblies previously reported for the each *Lenin* reactor. As such, added credence is given to the *Lenin* core history information.

**Table 4. Naval reactor core information for the former Soviet Union icebreaker *Lenin*.<sup>5,6</sup>**

Number of assemblies per reactor: 219		<sup>235</sup> U enrichment range: 4.6% to 6.4%	
Core Histories			
Naval Reactor	<sup>235</sup> U Loading (kg)	Operating Period (MW hours)	Effective Full Power Hours (hours)
1	80	560,000	8,600
2	76	550,000	8,500
3	129	660,000	10,000

For national security assets such as nuclear powered submarines, core history information like that published on the *Lenin* is virtually impossible to obtain. As such, a method for estimating the necessary reactor fuel load had to be developed. Assuming one knows the operating characteristics of the submarine, estimates of the reactor fuel load can be made from the power requirements of the submarine. For a submarine to operate at a given speed,  $S_i$ , the power requirement,  $P_i$ , in MW, is given by:

$$P_i = (\text{SHP}) (CF_1) (S_i / S_{\max})^3$$

where

SHP = shaft horsepower, hp, and  
 $CF_1 = 0.7457 \times 10^{-3} \text{ MW/hp}$ .

The overall power requirement of the reactor,  $P_R$ , in MW, is given by:

$$P_R = [(P_i / \text{PE}) + \text{HL}] / N_R$$

where

PE = propulsion efficiency,  
 HL = "hotel" load requirements, MW, and  
 $N_R$  = numbers of reactors.

The propulsion efficiency is that of the plant, and includes both thermal and mechanical conversion. The "hotel" load represents the total thermal power requirements of the submarine for all electric power and steam loads.

The minimum quantity of  $^{235}\text{U}$  required to power the submarine for a specific duration,  $^{235}\text{UL}_{\min}$ , in grams, is given by:

$$^{235}\text{UL}_{\min} = (CF_2) (P_R) (\text{AST}) (\text{CL})$$

where

$CF_2 = 1.24 \text{ grams } ^{235}\text{U/MWd}$ ,  
 AST = at-sea time, d/y, and  
 CL = core life, y.

The minimum quantity of U in the submarine reactor fuel load,  $\text{UL}_{\min}$ , in grams, is given by:

$$\text{UL}_{\min} = ^{235}\text{UL}_{\min} / E_U$$

where

$E_U$  = enrichment of  $^{235}\text{U}$ .

Note that the minimum quantity of U in the reactor fuel load,  $\text{UL}_{\min}$ , is not the amount that is actually predicted to be loaded in the submarine, but rather the minimum quantity of U required for the submarine to operate at speed  $S_i$  for a time period equal to the product of the at-sea time and core life. A substantially greater amount of U would be required for a full reactor load.

Table 5 presents a summary of the basic data used to estimate the minimum quantity of U in the reactor fuel load for each submarine whose discarded naval reactors contained SNF. Summarized for each of the various parameters is the range of values and the value assumed. The average speed at which each submarine was assumed to operate was arbitrarily set at 11 knots. For the shaft horsepower and maximum speed of the submarines, the average of the range of values was assumed. In the case of the propulsion efficiency, "hotel" load, at-sea time, and core life, the value assumed was the range limit or value that

would maximize the minimum quantity of U in the reactor fuel load. The value limits on enrichment are a best estimate from the available data. While the lower range limit is considered nominal for first-generation submarines of the November and Hotel class, the inclusion of a Yankee II class submarine requires the assumption of a range in enrichment.

**Table 5. Summary of the basic data used to estimate the minimum quantity of U in the reactor fuel load for each former Soviet Union submarine whose discarded naval reactors contained SNF.**

Parameter	Value Range	Value Assumed
November class SHP ( $10^3$ hp) <sup>7,8</sup>	30.0 - 35.0	32.5
Hotel class SHP ( $10^3$ hp) <sup>8,9</sup>	29.5 - 30.0	29.75
Yankee II class SHP ( $10^3$ hp) <sup>7,8,10</sup>	29.5 - 45.0	37.25
November class $S_{max}$ (knots) <sup>7,8</sup>	28 - 30	29
Hotel class $S_{max}$ (knots) <sup>8,9</sup>	23 - 26	24.5
Yankee II class $S_{max}$ (knots) <sup>7,8,10</sup>	26.5 - 27	26.75
Propulsion efficiency, PE, (%) <sup>11</sup>	15 - 20	15
Hotel load, HL, (MW)	12 - 15	15
Number of reactors, $N_R$ <sup>8,9,10</sup>	2	2
At-sea time, AST (d/y)	120	120
Core life, CL, (y)	5 - 7	7
<sup>235</sup> U enrichment, $E_U$ , (%) <sup>12</sup>	10 - 36	10 - 36

The radionuclide inventory in the SNF of the discarded naval reactors was calculated with ORIGEN2,<sup>13</sup> a point (no spatial dependence) depletion personal computer code that has been used extensively to characterize spent nuclear fuel and high level waste. The ORIGEN2 fixed data library used in these estimates is that for a generic PWR fueled with  $UO_2$  enriched to 4.2% in <sup>235</sup>U at a burnup of 50,000 MW days per metric tonne of U. A number of factors were considered in the selection of this particular library. First, 14 of the 16 discarded naval reactors are believed to be of the PWR type. Second, since the *Lenin* fuel matrix was described in the Yablokov Commission report as  $UO_2$ , it follows that the fuel matrix in first-generation submarine naval reactors built during the same period of time was also very likely  $UO_2$ . Third, the lowest <sup>235</sup>U enrichment in the *Lenin* reactors was quite close to 4.2%.

The highest <sup>235</sup>U enrichment considered for the submarines is substantially greater than 4.2%. One might expect that as the <sup>235</sup>U enrichment is increased, there will be a proportional decrease in the production of actinides. This is not the case; as the <sup>235</sup>U enrichment is increased, the neutron energy spectrum can be expected to harden or shift toward higher energies. With this shift in neutron spectrum, more resonance absorptions are expected to occur, which, in turn, will lead to a relative increase in the production of actinides. For a <sup>235</sup>U enrichment of 36%, the use of ORIGEN2 may result in an underestimate of the actinides by as much as a factor of two. The effect of a <sup>235</sup>U enrichment of 36% on the ORIGEN2 fission product estimate is believed to be significantly less. A more accurate estimate of the actinides in the higher enrichment fuels may be calculated with the computer code ORIGEN-S.<sup>14</sup> However, to perform this calculation, one must know either the relative shape and magnitude of the neutron energy spectrum or the composition and dimensions of a reactor fuel assembly or unit cell. Since information such as this was not readily available, the limitation in the prediction of the actinide inventory associated with the use of ORIGEN2 was considered acceptable.

To predict the activation product inventories in the reactor components and primary system corrosion products of the discarded naval reactors, the results of a British calculation for a generic nuclear powered submarine one year after shutdown were used.<sup>15,16</sup> Table 6 presents a summary of the British results. Summarized for each of the selected activation products are the radionuclide half-life, reactor component radioactivity, and primary system corrosion product radioactivity. Since the reactor power level of a

typical first-generation British submarine is similar to Western estimates of the reactor power levels of the discarded naval reactors, it follows that the data of Table 6 may be used without exception. For the reactor components the estimated total radioactivity is 79,100 Ci, with  $^{55}\text{Fe}$ ,  $^{60}\text{Co}$ , and  $^{63}\text{Ni}$  as the most dominant radionuclides, respectively. For the primary system corrosion products, the estimated total radioactivity is reduced to 111 Ci, with  $^{60}\text{Co}$  as the most dominant.

**Table 6. Information used to predict the radionuclide inventory in the reactor components and primary system corrosion products in the former Soviet Union naval reactors dumped in the Kara Sea.<sup>15,16</sup>**

Nuclide	Half-life (y)	Activity (Ci)	
		Reactor Components	Primary System Corrosion Products
$^{60}\text{Co}$	5.27	$1.27 \times 10^4$	$1.09 \times 10^2$
$^{14}\text{C}$	5,730	$1.14 \times 10^1$	$1.57 \times 10^{-5}$
$^{63}\text{Ni}$	100.1	$5.22 \times 10^3$	$2.61 \times 10^{-1}$
$^{55}\text{Fe}$	2.73	$6.11 \times 10^4$	$1.94 \times 10^0$
$^{59}\text{Ni}$	75,000	$4.68 \times 10^1$	$1.37 \times 10^{-3}$
Total		$7.91 \times 10^4$	$1.11 \times 10^2$

### Results

The maxima and minima in the estimated inventory of radionuclides presented herein were developed through an assessment of the variability of two key parameters:  $^{235}\text{U}$  enrichment and time between reactor shutdown and disposal of the SNF. The effect of  $^{235}\text{U}$  enrichment on the estimated inventory of radionuclides was evaluated for the *Lenin* and submarine naval reactors in the following way. In the case of the SNF from one of the three *Lenin* naval reactors, the reported range in  $^{235}\text{U}$  enrichment was assumed to be associated with a single three-reactor core load. Under a further assumption that the three *Lenin* reactors were loaded with approximately equal quantities of U, the  $^{235}\text{U}$  enrichments of 4.6% and 6.4% were associated with the two reactors loaded with 76 and 80 kg of  $^{235}\text{U}$  and the one reactor loaded with 129 kg of  $^{235}\text{U}$ , respectively. In the case of the six submarine naval reactors containing SNF, the assumed minimum and maximum in  $^{235}\text{U}$  enrichment were associated with separate reactor core loads.

The effect of time between reactor shutdown and disposal, or decay time, on the estimated inventory of radionuclides was evaluated by assuming a minimum decay time and a best estimate decay time for each naval reactor and disposal site. By definition, the minimum decay time for each naval reactor was chosen such that the estimate of the inventory of radionuclides at the time of disposal would be a maximum, and the best estimate decay time for each naval reactor was chosen such that a more realistic estimate of the inventory of radionuclides at the time of disposal would result. Table 7 presents a summary of the assumed time periods between reactor shutdown and disposal for the naval reactors dumped in the Kara Sea. Summarized for each disposal site is the disposal date, the number of discarded naval reactors and their associated ship identification number, the minimum decay time, and the best estimate decay time. With the exception of the two naval reactors that were discarded in Stepovoy Inlet in 1981, the minimum decay times were based on the Yablokov Commission finding of a minimum period of one year between reactor shutdown and disposal. The two naval reactors discarded in Stepovoy Inlet were earlier identified with the K-27, an assumed November class submarine that suffered a reactor accident on May 24, 1968. As such, their minimum decay time was established at thirteen years.

**Table 7. Assumed time periods between reactor shutdown and disposal for the former Soviet Union naval reactors dumped in the Kara Sea.**

Disposal Site	Disposal Date	Naval Reactors Discarded	Minimum Decay Time (y)	Best Estimate Decay Time (y)
Abrosimov Inlet	1965	2 (No. 285)	1.0	3.0
		2 (No. 901)	1.0	4.0
		2 (No. 254)	1.0	1.0
	1966	2 (No. 260)	1.0	1.0
Tsivolka Inlet	1967	3 (OK-150)	1.0	2.0
Novaya Zemlya Depression	1972	1 (No. 421)	1.0	4.0
Stepovoy Inlet	1981	2 (No. 601)	13.0	13.0
Techeniye Inlet	1988	2 (No. 538)	1.0	1.0

The best estimate decay time for each submarine whose discarded naval reactors contained SNF was assumed to be the time period, in whole years, between their associated accident and disposal dates. For those submarines whose discarded naval reactors are without SNF, the best estimate decay time was arbitrarily established at one year. In the case of the *Lenin*, whose reactor accident was reported to have occurred either early or late in 1966, the best estimate decay time was established at two years.

For the time of disposal, the inventories are estimated at 17 to 66 kCi of actinides plus daughters and 1,695 to 4,782 kCi of fission products in the SNF, 917 to 1,127 kCi of activation products in the reactor components, and 1.4 to 1.6 kCi of activation products in the primary system corrosion products. Our estimate of 1,695 to 4,782 kCi of fission products compares favorably with the Yablokov Commission finding of 2,300 kCi of fission products. In addition, of the 917 to 1,127 kCi of activation products in the reactor components, 161 to 184 kCi are associated with the  $^{60}\text{Co}$  inventory in the 16 discarded naval reactors. On a per-reactor basis, the estimated  $^{60}\text{Co}$  inventory in the reactor components is in excellent agreement with the Yablokov Commission finding of 100 kCi in the reactor components of ten naval reactors. With respect to the selected actinides and fission products, the disposal sites with greatest total activity are Tsivolka Inlet, the location of the *Lenin* remnants, and Abrosimov Inlet, respectively. For the activation product inventories in the reactor component and primary system corrosion products, the disposal sites with greatest total activity are Abrosimov Inlet and Tsivolka Inlet, respectively. Since the radioactivity in the reactor components and primary system corrosion products at a given disposal site is simply a function of the number of reactors discarded, it follows that Abrosimov Inlet should be the site of greatest activity.

Table 8 presents a summary of the estimated radioactivity in the SNF at the present time (1993). Summarized for each of the selected actinides and fission products are the minimum and maximum in radioactivity associated with the five disposal sites. With respect to the selected actinides, the radionuclide and disposal site with the greatest activity remain  $^{241}\text{Pu}$  and Tsivolka Inlet, respectively. With respect to the selected fission products, the radionuclides with greatest activity are now  $^{137}\text{Cs}$  and  $^{90}\text{Sr}$ , respectively. The disposal sites with greatest total activity are now Abrosimov Inlet and Stepovoy Inlet, respectively. Overall, for the present time (1993), the inventories are estimated at 6 to 24 kCi of actinides plus daughters and 492 to 540 kCi of fission products.

Table 9 presents a summary of the estimated radioactivity for selected activation products in reactor components and primary system corrosion products at the present time (1993). Summarized for each of the selected activation products are the minimum and maximum in radioactivity associated with the five disposal sites. With respect to the reactor components, the radionuclides with greatest activity are  $^{60}\text{Ni}$  at



**Table 8. Estimated radioactivity in the SNF at the present time (1993) for the former Soviet Union naval reactors dumped in the Kara Sea.**

Nuclide	Disposal site activity range (Ci)										
	Abrosimov Inlet	Tsivolka Inlet	Novaya Zemlya Depression	Stepovoy Inlet	Techeniye Inlet	All sites					
<i>Actinides</i>											
<sup>239+240</sup> Pu	94	474	343	374	33	167	55	280	--	526	1,295
<sup>241</sup> Am	14	605	412	688	5	204	6	277	--	436	1,774
<sup>238</sup> Pu	18	258	102	148	7	105	9	123	--	136	634
<sup>241</sup> Pu	117	5,710	4,450	7,690	61	2,990	79	3,510	--	4,707	19,900
Subtotal	243	7,047	5,307	8,900	106	3,466	149	4,190	--	5,805	23,603
All	247	7,050	5,310	8,900	108	3,450	152	4,190	--	5,817	23,590
<i>Fission Products</i>											
<sup>129</sup> I	0.02	0.03	0.01	0.01	0.009	0.009	0.01	0.02	--	0.06	0.06
<sup>90</sup> Sr	46,200	51,400	18,500	23,700	19,000	21,400	32,100	33,400	--	115,800	129,900
<sup>134</sup> Cs	0.3	3	2	3	1	12	2	6	--	5	23
<sup>137</sup> Cs	51,300	54,600	21,900	26,800	21,100	22,600	35,400	35,400	--	129,700	139,400
<sup>154</sup> Eu	152	190	87	116	98	125	132	132	--	469	563
<sup>125</sup> Sb	2	6	5	6	4	12	7	9	--	18	33
<sup>147</sup> Pm	63	155	72	119	128	345	225	365	--	488	984
<sup>155</sup> Eu	31	49	27	33	28	45	47	51	--	133	178
<sup>99</sup> Tc	15	15	6	7	5	9	9	--	--	34	36
<sup>151</sup> Sm	686	1,360	271	380	496	468	866	--	--	1,669	3,102
Subtotal	98,449	107,777	40,869	51,164	40,608	45,040	68,389	70,238	--	248,316	274,220
All	195,000	213,000	80,900	101,000	80,500	86,700	136,000	139,000	--	492,400	539,700

**Table 9. Estimated radioactivity of selected activation products in the reactor components and primary system corrosion products at the present time (1993) for the former Soviet Union naval reactors dumped in the Kara Sea.**

Disposal site activity range (Ci)							
Nuclide	Abrosimov Inlet	Tsivolka Inlet	Novaya Zemlya Depression	Stepovoy Inlet	Techeniye Inlet	All sites	
<i>Reactor Components</i>							
<sup>60</sup> Co	2,297	1,100	542	1,080	13,200	18,219	18,988
<sup>14</sup> C	91	34	11	23	23	182	182
<sup>63</sup> Ni	34,140	13,000	4,420	8,480	10,100	70,140	70,610
<sup>55</sup> Fe	336	193	138	276	34,300	35,243	35,550
<sup>59</sup> Ni	374	140	47	94	94	749	749
All	37,238	14,467	5,158	9,953	57,717	124,533	126,079
<i>Primary System Corrosion Products</i>							
<sup>60</sup> Co	20	9	5	9	113	156	162
<sup>14</sup> C	0.0001	0.00005	0.00002	0.00003	0.00003	0.0002	0.0002
<sup>63</sup> Ni	1.7	0.6	0.2	0.4	0.5	3.5	3.5
<sup>55</sup> Fe	0.01	0.006	0.004	0.009	1.1	1.1	1.1
<sup>59</sup> Ni	0.01	0.004	0.001	0.003	0.003	0.02	0.02
All	21	10	5	10	115	161	167

Abrosimov Inlet and  $^{55}\text{Fe}$  at Techeniye Inlet, while the disposal site of greatest activity is now Techeniye Inlet. With respect to the primary system corrosion products, the radionuclide and disposal site with greatest activity are  $^{60}\text{Co}$  and Techeniye Inlet, respectively. That Abrosimov Inlet is no longer the site of greatest activity is not surprising. While the radioactivity in the reactor components and primary system corrosion products at a given disposal site remains a simple function of the number of reactors discarded, when radioactive decay of the activation products is considered, Techeniye Inlet becomes the expected site of greatest activity. Overall, for the present time (1993), the inventories are estimated at 125 to 126 kCi of activation products in the reactor components and 0.16 to 0.17 kCi of activation products in the primary system corrosion products.

### Assessing the Health Risks of Arctic Contamination

The process of assessing the risks of radioactive materials associated with naval reactors disposed in Arctic waters, as well as radionuclides, organic compounds, and metals derived from terrestrial sources, begins with analyses of the inventories and release rates of the contaminants of potential concern. The results of these analyses are then used as inputs to contaminant transport models that predict the concentrations of the substances in environmental media; for example, the marine sediments and water in a given ocean region. Food-chain transfer models are used subsequently to determine the movement of substances from the contaminants, the resulting doses to critical organs, and the estimation of associated health risks using dose-response relationships. During the early phases of a risk assessment, though, it is not always possible to assess rigorously all of the exposure pathways for various contaminants. In fact, it is often prudent to complete a screening-level analysis to identify the most important contaminants from a risk standpoint. An important benefit of such an exercise is that it can quickly eliminate from further consideration contaminants or exposure pathways that contribute little to health risk. The results of initial analyses can also be used to devise monitoring programs that focus on the high-risk contaminants and associated environmental media.

To demonstrate the use of a preliminary risk assessment, we have prepared an analysis of the collective population dose and fatal cancers resulting from the release of key actinides and fission products contained in the discarded SNF discussed earlier. We have used a series of dose-conversion factors that translate radioactivity released into an environmental compartment (e.g., the atmosphere or sea water) to a collective dose commitment. These radionuclide-specific factors are expressed as the ratios of collective dose commitment per unit of radioactivity released (units of person-Gy per Bq). The use of such factors originates from dose assessments of populations worldwide that have been exposed to mixed fission products released to the global environment via the atmosphere from nuclear tests and accidents. There is now considerable information on the regional- and global-scale transfers of key radionuclides from air, water, and soils to food-chains, making it possible to prepare estimates of radiation doses to the global population. As an example, Anspaugh, et al.<sup>18</sup>, in a detailed assessment of the Chernobyl accident, calculated a collective dose commitment of 930,000 person-Gy from the release of fission products. Assuming that the estimated dose is dominated by the 100 PBq of  $^{137}\text{Cs}$  emitted from the reactor, we can calculate a dose-conversion factor of  $9 \times 10^{-12}$  person-Gy/Bq released. In an unrelated study, the World Health Organization (WHO)<sup>19</sup> has derived a dose-conversion factor for  $^{137}\text{Cs}$  of  $5 \times 10^{-12}$  person-Gy/Bq released, a difference of only a factor of 1.8.

For our analysis, we used dose-conversion factors (DCFs) presented in the WHO study for collective doses commitments associated with unit (i.e., Bq) releases of radionuclides to sea water. The DCFs for each of the radionuclides considered were based on the discharge of one Bq of activity to sea water, subsequent dilution and radioactive decay, and food-chain transfers via the consumption of finfish and shellfish. Table 10 shows the DCFs for  $^{90}\text{Sr}$ ,  $^{241}\text{Am}$ ,  $^{137}\text{Cs}$ , and  $^{239}\text{Pu}$ , which range from  $0.047 \times 10^{-14}$  to  $6.4 \times 10^{-14}$  person Sv/Bq. When we multiply DCFs by the estimated inventories of the key radionuclides in the discarded SNF, we find that the highest collective dose commitments were associated with  $^{137}\text{Cs}$  and  $^{241}\text{Am}$ ,

respectively. It is interesting to note that the predicted inventory of  $^{90}\text{Sr}$  is much greater than that of  $^{241}\text{Am}$ ; however,  $^{241}\text{Am}$  has a higher collective dose commitment than  $^{90}\text{Sr}$  because the DCF of  $^{241}\text{Am}$  is much greater than that of  $^{90}\text{Sr}$ . The collective dose commitment calculated for  $^{239}\text{Pu}$  is much lower than for the other radionuclides, suggesting that it does not need to be the primary focus of more detailed assessments or monitoring efforts.

**Table 10. Estimate of global population dose for radioactive wastes in the Arctic Ocean.**

Nuclide	Inventory <sup>a</sup> (Bq)	Dose-conversion Factor <sup>b</sup> ( $10^{-14}$ Person-Sv/Bq)	Collective Dose Commitment (Person-Sv)
$^{90}\text{Sr}$	$4.8 \times 10^{15}$	0.047	2.3
$^{241}\text{Am}$	$6.6 \times 10^{13} \text{ }^c$	6.4	4.2
$^{137}\text{Cs}$	$5.2 \times 10^{15}$	0.1	5.2
$^{239}\text{Pu}$	$4.8 \times 10^{13}$	0.21	0.1

<sup>a</sup>Inventory for 1993.

<sup>b</sup>From WHO (1983) and includes both fish and shellfish.

<sup>c</sup>Includes growth from  $^{241}\text{Pu}$ .

A crude estimate of population risk can be made by multiplying the total collective dose equivalent by a cancer risk factor for ionizing radiation, or

$$\text{Population Risk} = (11.8 \text{ person} \cdot \text{Sv})(0.05/\text{Sv}) = 0.6 \text{ fatal cancers.}$$

By comparison, Anspaugh et al.<sup>18</sup> estimated that 17,000 fatal, radiation-induced cancers could occur as a result of the Chernobyl accident. This initial analysis of the discarded SNF indicates that it does not constitute a significant problem on a global scale. This is not too surprising because releases of radioactive contaminants to marine systems do not produce external radiation exposures, and unlike atmospheric releases, there is no widespread contamination of soils and crops. We should stress, though, that our results are sensitive to the turnover times of the radionuclides in receiving waters and the release rates of radioactive materials from the discarded SNF.

### Conclusions

Considering the uncertainties associated with certain of the analytical model parameters and in the times between reactor shutdown and disposal, the estimates presented herein agree quite favorably with the Yablokov findings for the time of disposal.

At the present time (1993), even if one assumes that the actinides are underestimated by a factor of two, the inventories of actinides and fission products in the SNF and the inventories of activation products in reactor components and primary system corrosion products are estimated to be no greater than 47 kCi, 540 kCi, 126 kCi, and 0.17 kCi, respectively. Total inventory is estimated at less than 713 kCi.

The preliminary assessment of the global population risk from release of the present (1993) activity levels of key actinides and fission products in the discarded SNF is estimated to be 0.6 fatal cancers.

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Cs-137 contamination of sea water surrounding the "Komsomoletz" nuclear submarine.

The state of the nuclear reactor plant of the "Komsomoletz" submarine lying on the Norway sea for since april 1989 is one of the main factors determining radioecologic situation in that region. Cs-137 is the most appropriate fission product for registration in sea water. This radionuclide's content in sea water surrounding sunk sub. was first measured by scientists of the Russian Research Center (RRC) "Kurchatov Institute" during an expedition works on board R&D ship "Academic Mstislav Keldysh", which were undertaken by the Central Design Bureau for Marine Engineering "Rubin" (Russia) in the period of time since august 10 till september 15, 1991.

Side by side with radiochemical methods an express one was proposed for measuring of Cs-137 concentration in sea water surrounding the submarine whereabouts. A submergible gamma-spectrometer on board the "Mir" manned submergence vehicle used for registration of gamma-radiation spectrums in sea water along the vehicle route. Subsequent analysis of the information gained allowed to fix the sites of pollution and to estimate Cs-137 concentration.

The measurements were made with the help of gamma-spectrometer REM-1, which was developed by RRC. The gamma-spectrometer consisted of a high sensitive scintillations detector mounted on the outer structures of the "Mir" submergence and a registration part of the complex inside the submergence. The two parts of the complex were connected by a three lines wiring.

Submergible detector was made on the base of monocrystal of sodium iodide with the diameter of 200 mm and the height of 200 mm which was optically joined with a photoelectronic multiplier having the diameter of 170 mm. All the parts of the detector were placed in a pressurized titanium capsule (See Fig.1.). Amplification of the output signals from the multiplier and their transformation into a digital code were made by an electronic unit of the module. Digital code were transferred through the buffer to the input of the submergible part of the complex in the form of a standard successive code. Power supply of 24 V DC was taken from the "Mir" submergence and then transformed with the help of secondary sources into voltages needed for the electronic unit and the multiplier.

The recording part of the complex was made on the base of

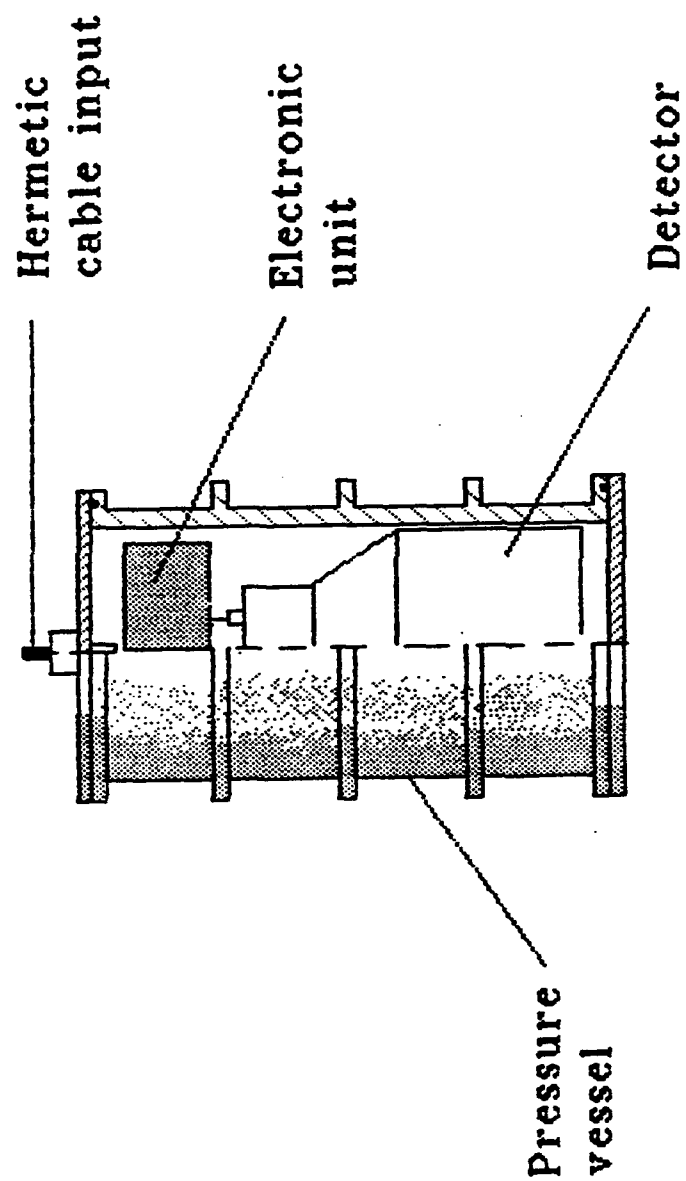


Fig.1 Gamma-spectrometer REM-1.



personal computer PC/AT type and consisted of a processor, a monitor and a key board. Digital codes were transferred from the submergable part of the complex through standard successive interface and then were received and processed by the computer.

Information accumulation program allowed to install expositions duration from 30 sec to 30 min, to make singular or cyclic measurements of gamma-spectrums and to mark current measurements with special marks.

Laboratory investigations of the complex parameters gave the next results:

- diapason of gamma-quantums energy registered: from 0.1 to 3.0 MeV;
- an integral non-linearity of the spectrometre's energy scale is lower than 1 % ;
- the resolving power of the spectrometer related to gamma-quantums from Cs-137 (0.66 MeV) - 10.5 % ;
- variations of a gain coefficient of photomultiplier in the Earth magnetic field are lower than 0.5 % .

The module's sensitivity measurement was made in a water tank having a capacity of 2.2 m. 75 ml of Cs-137 solution with the specific radioactivity of 8.52 10 Bq/g were added to water. So the concentration of Cs-137 in the tank was 7.8 nCi/l. The value of sensitivity in energy diapason 580 - 750 keV was obtained in the results of the experiment was  $0.08 \pm 0.02$  cps/pCi/l.

The gamma-spectrometer was placed on the "Mir" deep submergence vehicle and registered gamma-radiation spectrums caused by water radioactivity and in some cases by radioactivity of bottom sediments. Exposition during cycling registrations of spectrums was chosen to be equal of 3 min.

192 of spectrums were registered during the first submergence of the vehicle.

Spectrums caused only by natural radioactivity of sea water (that is registered at a distance more than 1.5 - 2.0 m from the bottom or from the hull of the sub) were as a consequence of K-40 gamma-radiation and did not contain anomalies (Cs-137 traces included). One can see a typical spectrum registered on the Fig.2. In the cases of more noticeable influence of bottom sediments radiation a common counts level of the spectrometer rose in the all partes of gamma-radiation energy distribution. (See Fig.3) There were no exceedings of counts rate registered caused by photoabsorption of Cs-137 gamma-radiation.

A row of spectrums was registered while placing the detector in the direct vicinity to the hull of sub at the sites of the most probable output of Cs-137 from inside the hull. As a result of initial processing of that measurements serial the spectrums were

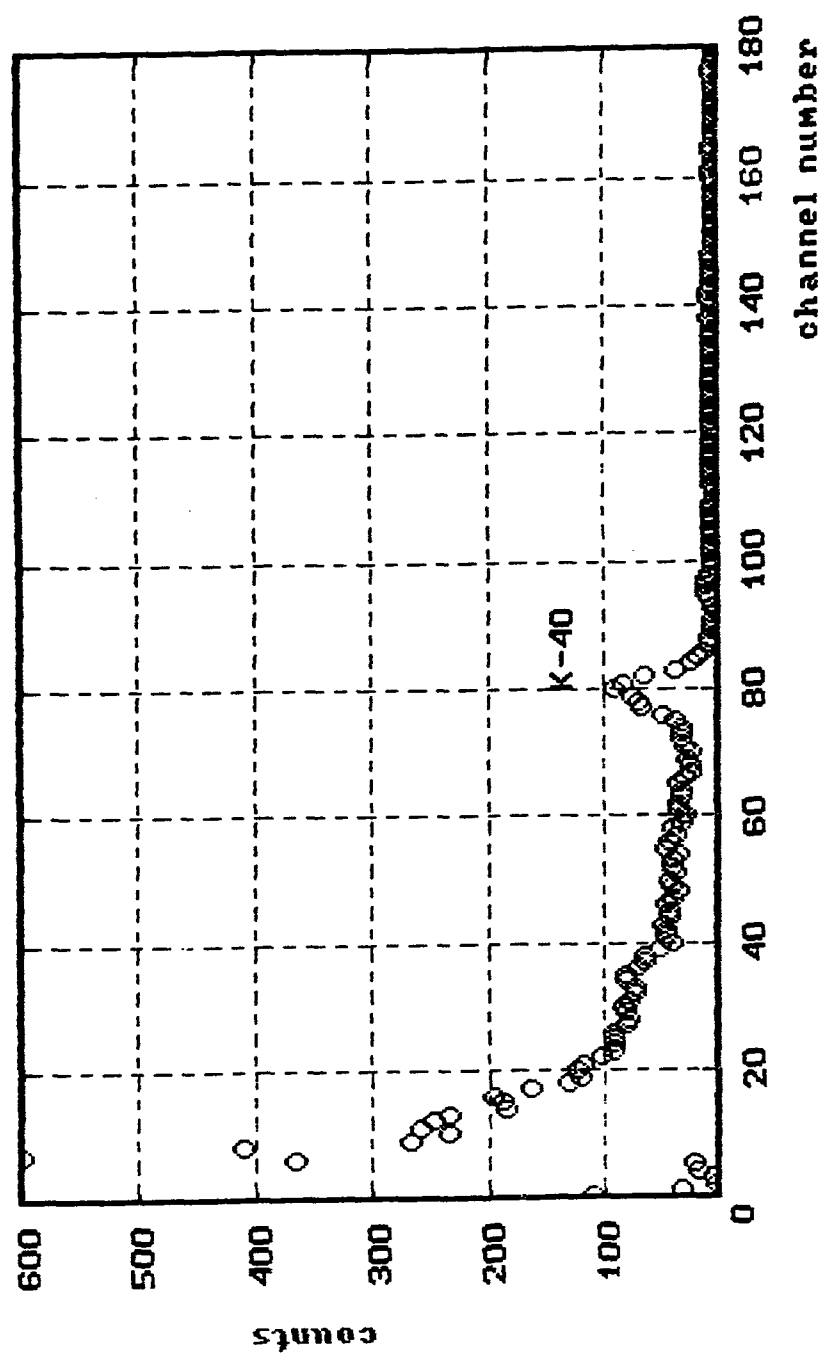


Fig.2 Typical gamma-spectrum of sea water.

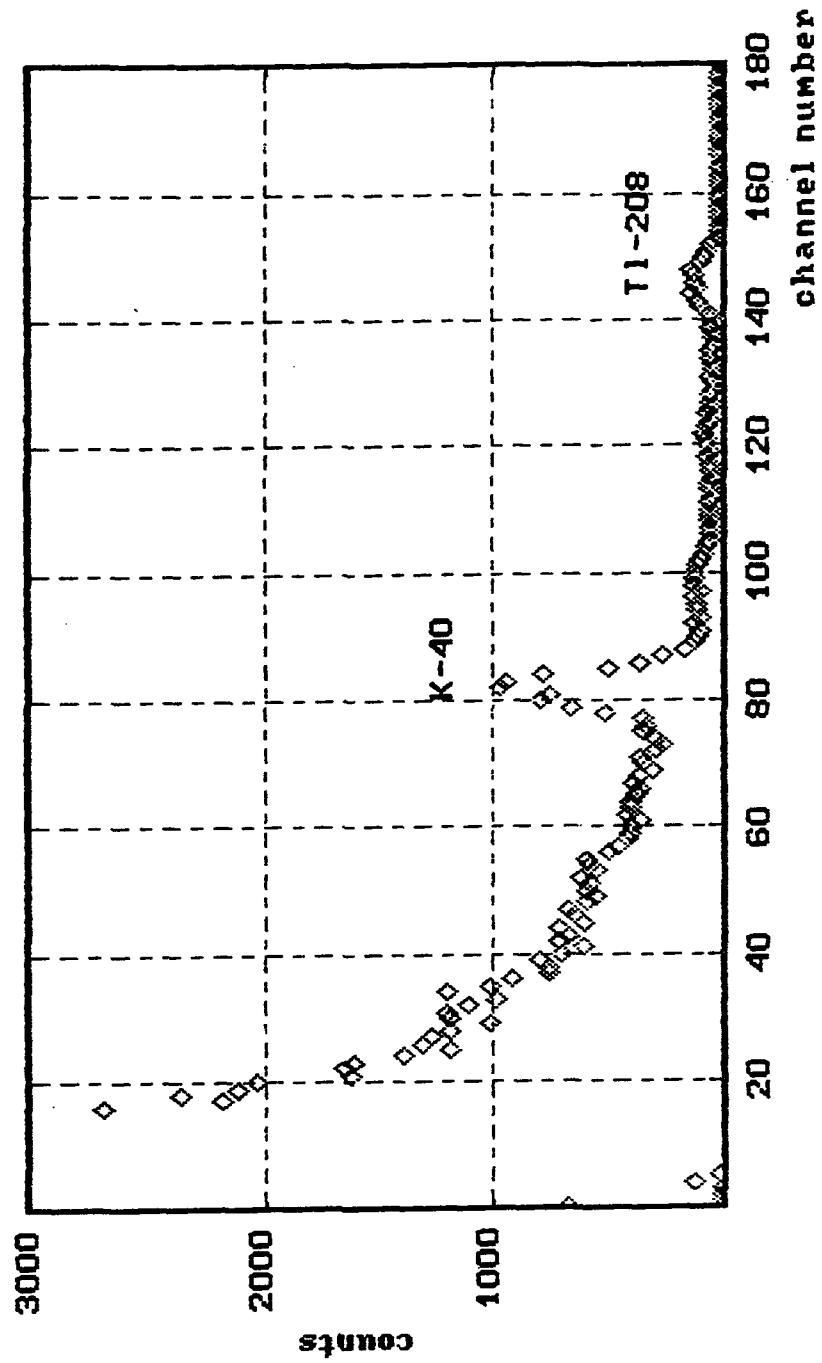


Fig.3 Typical gamma-spectrum of sediments.

discovered which have an anomaly form in 0.66 MeV region (See Fig. 4) Basing on the spectrometer sensitivity it is possible to estimate Cs-137 concentration at that point of the spectrum as  $0.11 \pm 0.03$  nCi/l. This spectrum corresponds to the position of the deep submergence vehicle in the vicinity to the nuclear reactor compartment of the sub.

Measurement were made during the second submergence of the "Mir" vehicle on August 25, 1991 in the same way like in the first attempt. 192 spectrums were registered with the exposition time of 3 min. No spectrums with Cs-137 traces were found after processing. 215 spectrums were registered during the third submergence of the vehicle on August 31, 1991. Analysis of them confirms Cs-137 presence at the same point as in the first submergence. Those spectrums are shown on the Fig. 5. It could be seen that 20 min before Cs-137 registration the deep submergence vehicle was at the position on the sea floor, because of K-40 and Tl-208 piks presence in the spectrum caused by bottom sediments radiation. The form of spectrum N45 allows to suppose gamma-lines of Cs-134 (605 keV and 795 keV) presence in it along with Cs-137 lines. Subsequent analysis of the spectrum included its decomposition according to Cs-137, Cs-134 photopiks and background. The last one was approximated by exponent. The results of such an interpretation of the spectrums obtained are shown on the Fig. 6.

Supposing the sensitivity of the spectrometer equal to  $80 \pm 20$  cps/nCi/l it is possible to estimate Cs-137 concentration of  $0.6 \pm 0.2$  nCi/l.

Thus in its three submergences in 1991 for surveying the sunk nuclear submarine "Komsomolets" the deep submergence vehicle "Mir" had registered the presence of Cs-137 only at one of the sites: at close proximity to pivoted ventilation window of the sub's reactor compartment. Cs-137 concentrations at that site was measured as  $0.11 \pm 0.03$  nCi/l on August, 25 and as  $0.6 \pm 0.2$  nCi/l on August, 31 measurements.

There were no Cs-137 registrations at all other sites with concentrations more than sensitivity limits of the spectrometer at exposition time of 3 min., that is - 3 pCi/l. Those investigative measurements have put the starting point of radioecologic situation monitoring in the vicinity to the sunk nuclear submarine the "Komsomolets" whereabouts.

The next step in continued systematic investigation of a submarine reactor dynamic state under guidance of Central Design Bureau of Marine Technology "Rubin" was in 1992, when works of the new expedition were conducted. The program of the works included measurements like those made in 1991 as well as investigations of sea water pollution at sites remoted from the sunk sub by several

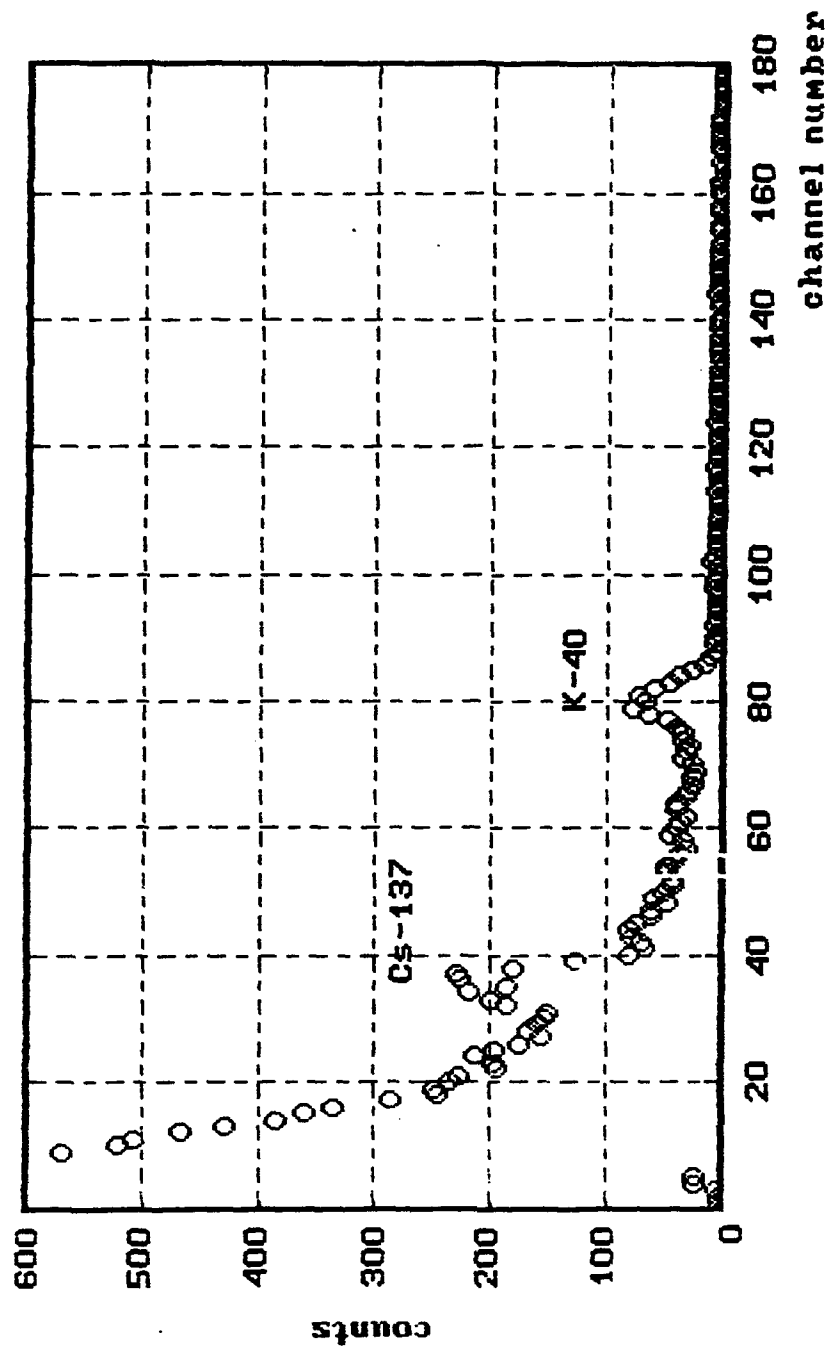


Fig.4 Gamma-radiation spectrum of water in the vicinity to reactor compartment.

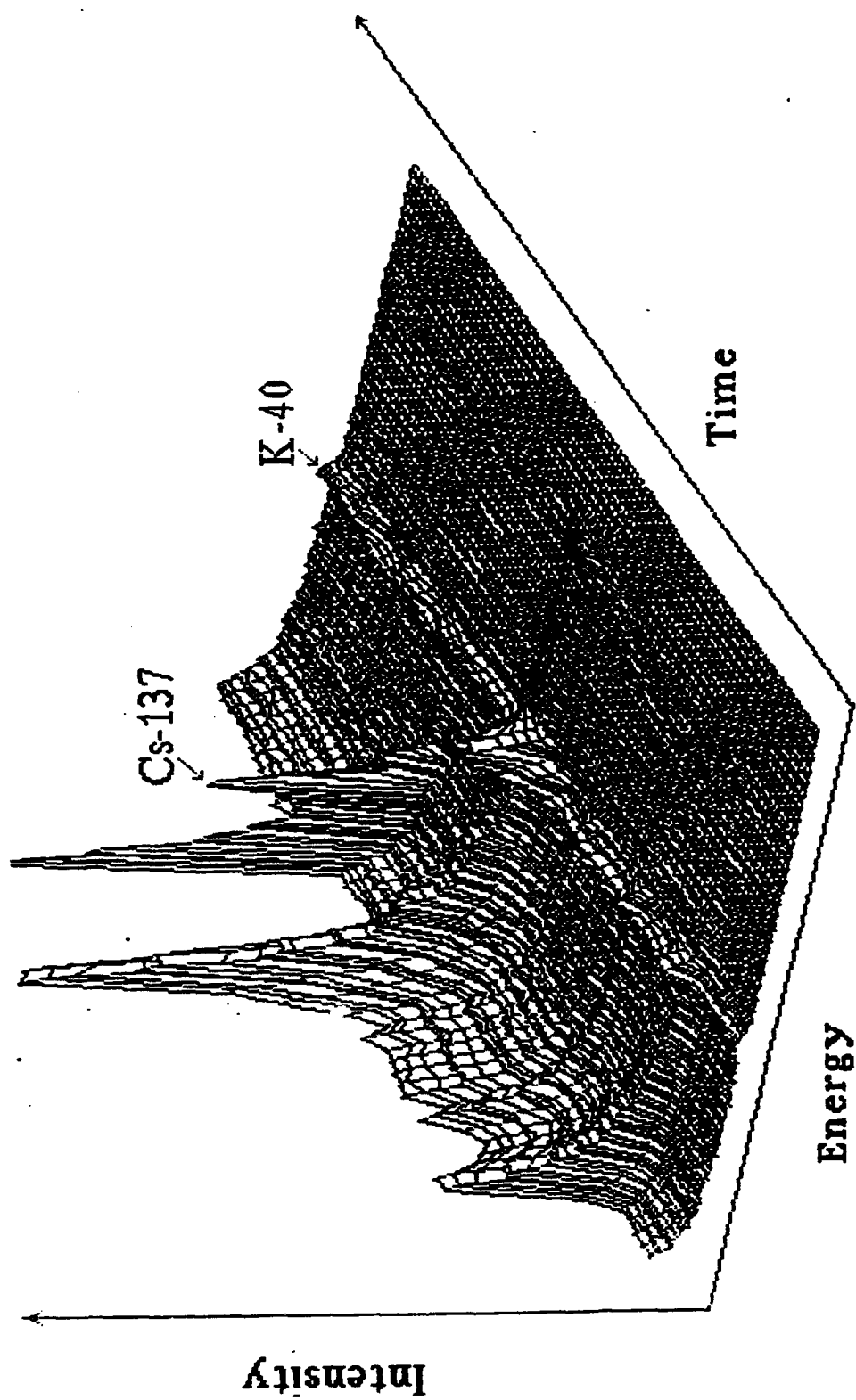


Fig.5 Radiation spectra composition changes in 240 min. of spectrometers work.

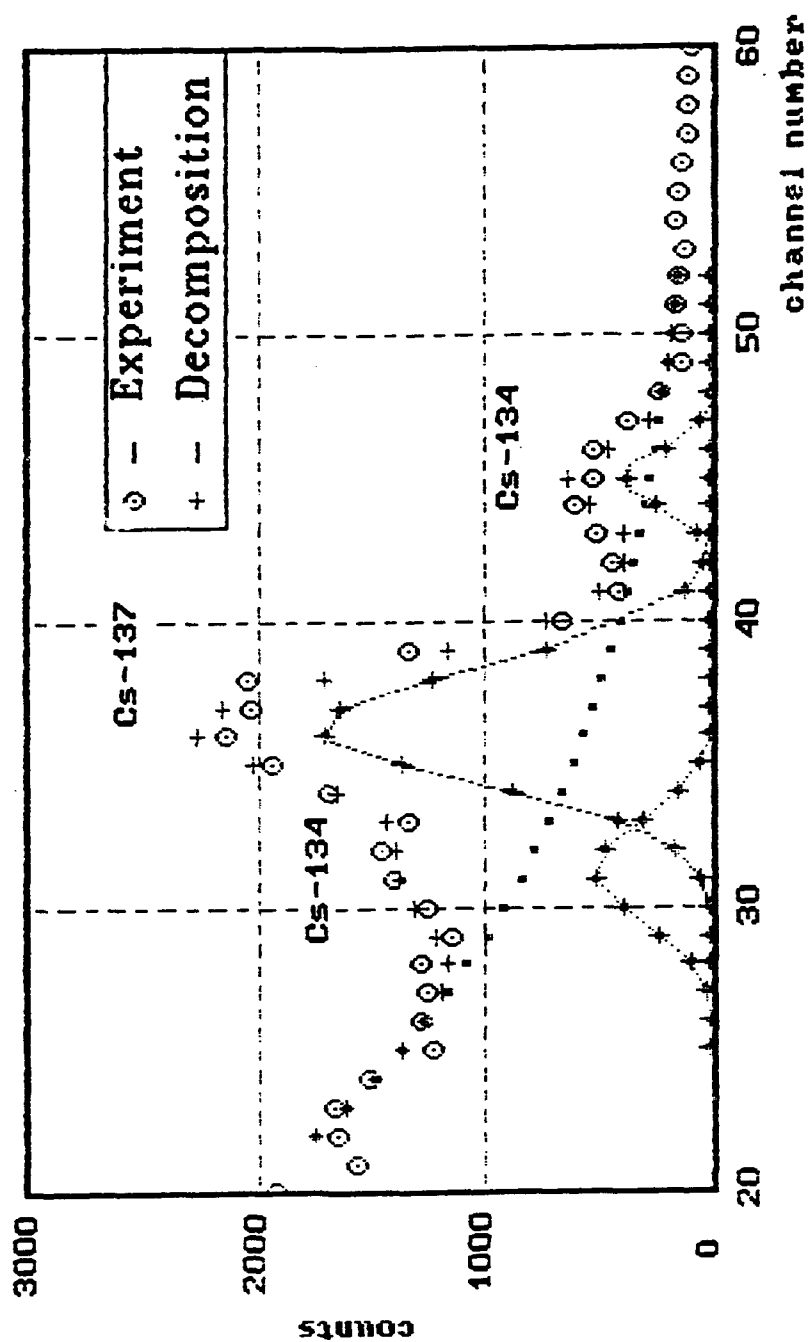


Fig.6 Decomposition of experimental spectrum.

miles.

There had been developed and manufactured a gamma-spectrometer REM-2 particularly for Cs-137 measurements in a close vicinity to submarine (not farther than 10m from submarine the hull). The gamma-spectrometer REM-1 used in 1991 year measurements served as a prototype unit for a new gamma-spectrometer (see Fig.7). The gamma-spectrometer REM-2 consists of a high sensitive scintillation detector and a registration part of the complex. The detector is made on the base of sodium iodide monocrystal with a diameter of 200 mm and a height of 100 mm. It is enclosed in a spheric pressure vessel (capsule) made of a high strength aluminium alloy. The spheric submergible capsule of the detector is mounted on the outer structures of the deep submergence vehicle. The registration part of the complex is placed inside a vehicle.

Despite of the facts that the REM-2 gamma-spectrometer has a crystal of less dimensions and correspondingly less sensitive detector than REM-1 used in 1991 year it is more appropriate for work in deep submergence vehicle conditions. Weight of a submergible part of the module in the air is 30 kg (weight of the REM-1 is 80 kg) and its weight in water is 10 kg (REM-1 weighs 25 kg). Measurements of sea water radioactivity in the vicinity to the "Komsomolets" submarine were made during submergences of the "Mir-2" vehicle on the 23-th, 24-th, 25-th and 29-th of May, 1992. Methodology of the measurements was the same as in 1991 year with the REM-1 gamma-spectrometer. The detector unit of the REM-2 gamma-spectrometer for work periods was mounted above the left support of the vehicle "Mir-2". The gamma-spectrometer registered gamma-radiation spectrums caused by radioactivity of sea water and bottom sediments along the routs of the vehicle movements. Exposition time during cycling spectrums registrations was chosen of 3 min.

The whole quantity of the spectrums registered was 171. There had been registered insignificant rise of spectrometer counts rate above the background in an energy diapason from 0.5 Mev to 0.8 Mev during the vehicle positioning above the reactor compartment of the submarine. It had been no chances for estimation of Cs-137 concentration on the base of singular measurement results. That is why the attempt of sum total of the spectrums gaining was made for the whole time interval of the vehicle being positioned above the reactor compartment. Thus mean values of Cs-137 concentration for the reactor compartment area was estimated as  $0.8 \pm 0.2$  pCi/l. Registration of Cs-137 was made at the same point in 1991 by the REM-1 gamma-spectrometer. Processing of the spectrums registered at the other sites of the "Mir-2" vehicle positionings has not uncovered Cs-137 contents concentrations exceeding



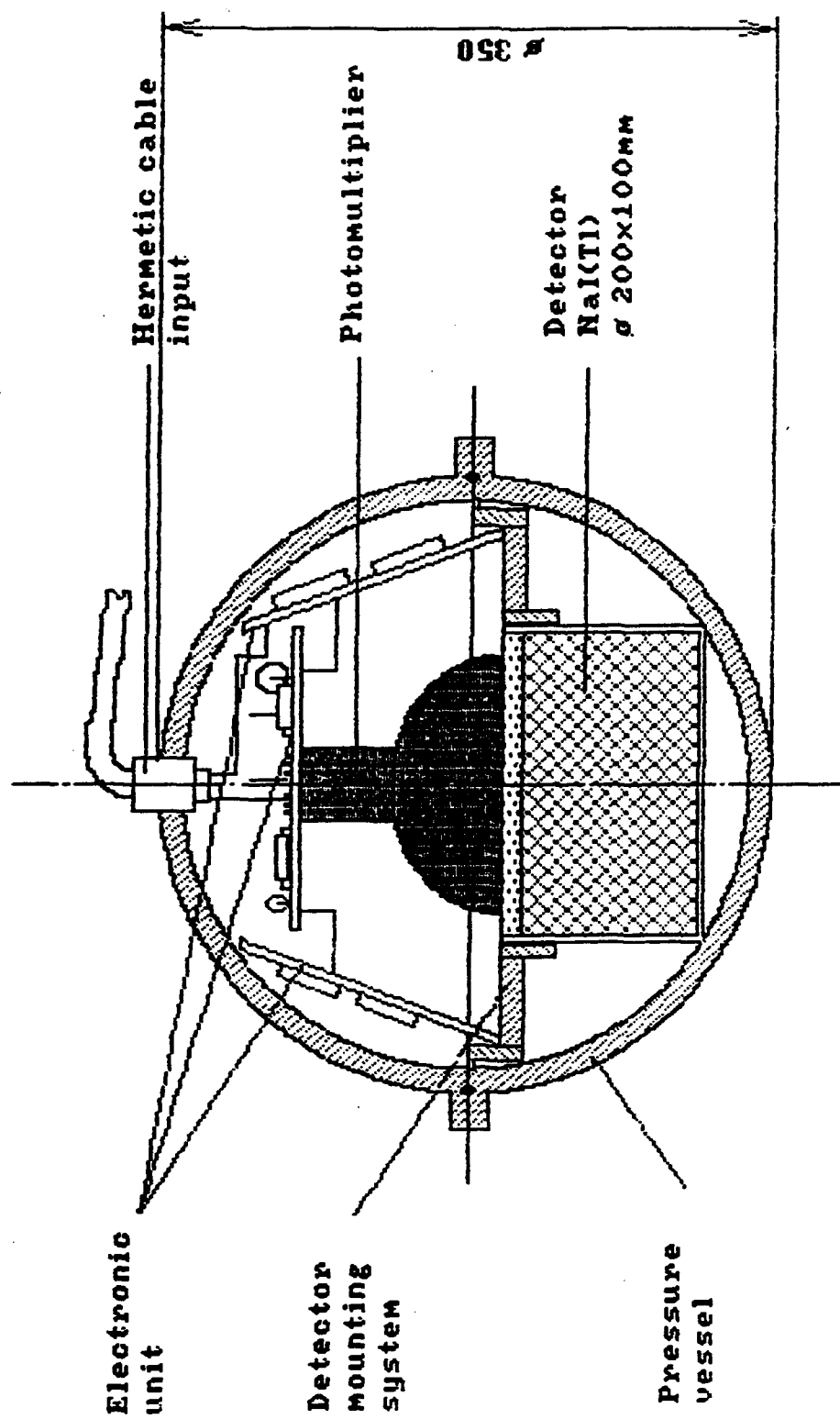


Fig.7 Gamma-spectrometer REM-2.

sensitivity limits of the gamma-spectrometer ( $1\text{pCi/l}$ ). There were measurements made during the second and the third submergences of the "Mir-2" vehicle following the same methodology as at the first submergence, the area of the reactor compartment included. There had been registered 77 spectrums at the second submergence and 161 spectrums at the third one. There was no Cs-137 concentration exceeding  $1\text{pCi/l}$  uncovered after those spectrums processing. The results obtained could be explained by decreasing of Cs-137 output from the hull of the sub. Thus the conservatism of the radiation situation in the area of the "Komsomolets" whereabouts was confirmed.

A REM-5 gamma-spectrometer had been developed for Cs-137 concentrations measurements in more wide area around the "Komsomolets". A submergable gamma-spectrometer "REM-5" consists of a unic scintillation detector made on the base of NaI(Tl) monocrystal about 12.5 l in volume which is joined with photomultiplier and an electronic unit. The last allows to transfer analogue spectrometric signals through a single wire cable of 500 m length to a shipborn unit. The signal there is transformed into a digital code and then transferred to a computer (IBM PC/AT type) for processing. An autonomous version of the electronic unit has been developed as well which allows to accumulate in a preprogrammed way up to 16 gamma-spectrums inside spectrometer memory.

A detector part of the spectrometer is enclosed in a strong titanium capsule. The module weight in air is 110 kg and in water - 60 kg. A sensitivity limit of the module as related to even distributed Cs-137 isotope in water is  $0,3\text{ pCi/l}$  while exposition time is 30 min.

A sounding of water down to depth about 2100 m was undertaken from the board of the R&D Ship "Academic Mstislav Keldysh" on the 21-st of May, 1992 with the help of the REM-5 gamma-spectrometer equipped with an autonomous version of its electronic unit.

16 gamma-spectrums had been registered with exposition time of 10 min. Spectrums numbers 4,5 and 6 were registered while submerging the probe with the downing speed about 2 m/s down to the maximum depth of 2100 m. The spectrum N7 was obtained on the maximum depth. Spectrums numbers from 8th to 12th were registered while lifting the probe. Processing of the spectrums registered in this experiment has not uncovered Cs-137 concentrations in sea water exceeding the sensitivity limit of the gamma-spectrometer which is  $0.5\text{ pCi/l}$ .

A high sensitive submergable gamma-spectrometer was used for experimental check up of a possibility of an express investigation of the degrees and structures of wide aquatorias pollution by Cs-137 isotope. With that objective a sounding of water was made at

three sites of the Baltic sea on the depth of 10 m down to the sea floor. The summary time of measurements at each depth was 15 min. One of the spectrums obtained is shown on Fig.9. Using data obtained a vertical profile of Cs-137 content in sea water was construed (See Fig.8). It could be seen from results shown that it is possible now to in-situ investigate a structure of Cs-137 content in water of the Baltic Sea.

Thus apparatus and methodologies for measurements of radioactivity of sources of marine environments radionuclides pollution have been developed and checked up. It is possible now to in-situ investigate, with the help of unic submergable gamma-spectrometric complexes, of radioactive isotope Cs-137 content in World Ocean's water down to concentration limit of 0.5 pCi/l. The methodology could be applicable to express analysis of radioecologic situation in sea aquatorias and for searching of sources of radionuclides pollution of marine environment.

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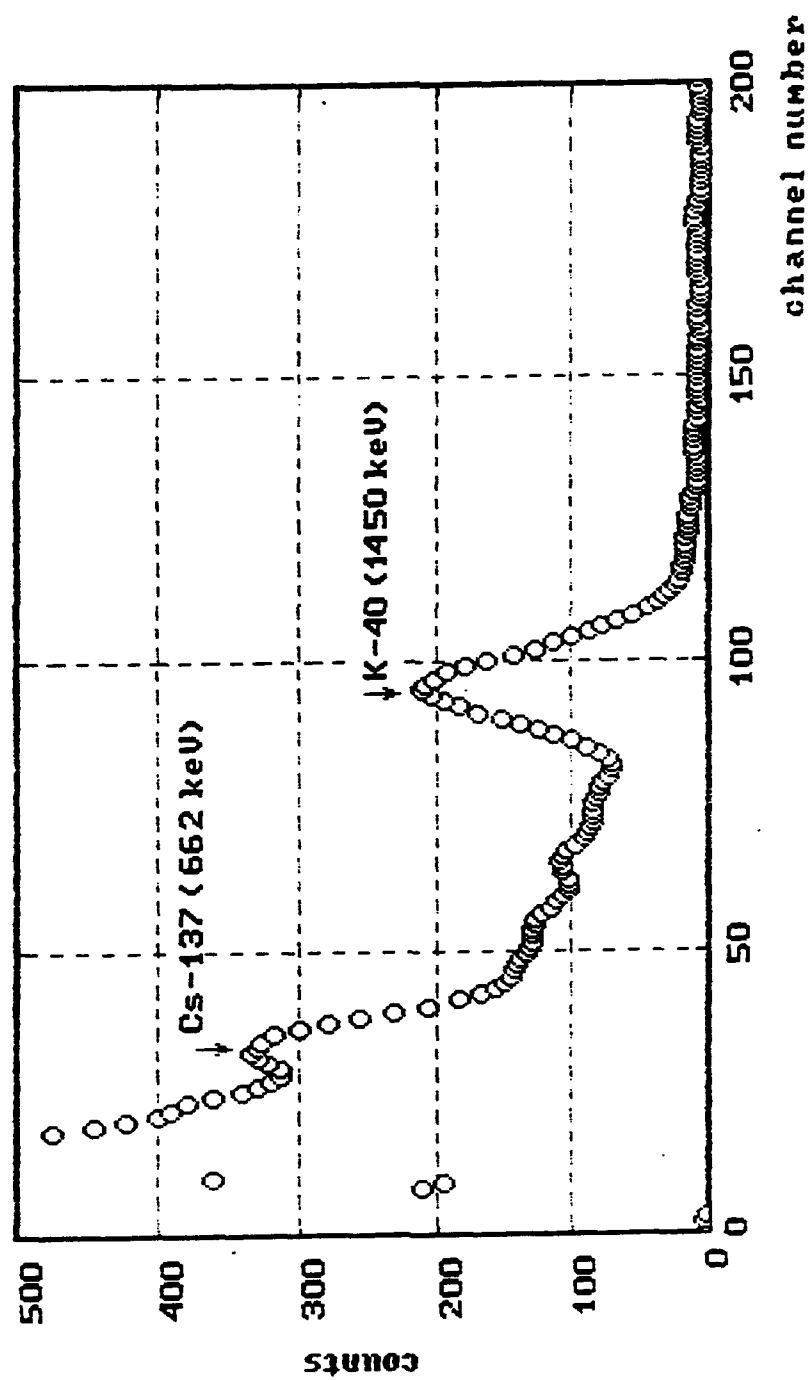


Fig.8 Gamma-spectrum of Baltic sea water.  
Depth 20 m.

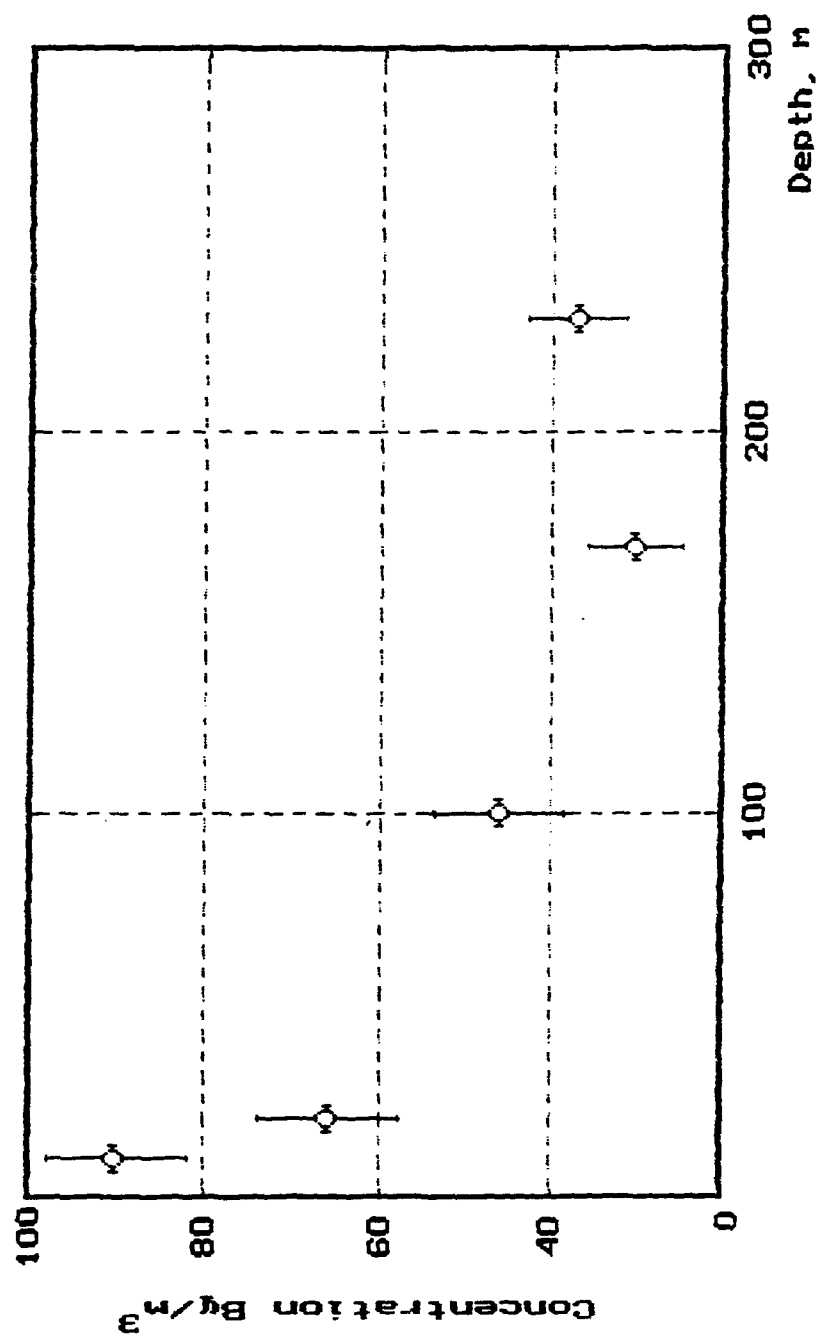


Fig.9 Vertical distribution of Cs-137 in Baltic sea.



POSTER TEXT WOODS HOLE 7-9 JUNE 1993

## BOMB RADIOCARBON IN ARCTIC ALASKAN AQUATIC AND TERRESTRIAL BIOTA

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Bomb radiocarbon from atmospheric weapons testing provides an effective tracer for biological processes in both marine and terrestrial environments. The sharp peak in  $^{14}\text{C}$  activities associated with atmospheric testing in 1961-63 can be used to follow food webs based on multiple carbon sources and to validate models of plant decomposition and estimate rates of modern soil carbon storage.

## INTRODUCTION

The atmospheric testing of nuclear weapons prior to 1963, equivalent to about 550 megatons of TNT, produced about 12 kg  $^{14}\text{CO}_2$  per megaton of nuclear explosive. In the northern hemisphere, this served to almost double the ambient atmospheric  $^{14}\text{C}$  concentration. Rapid atmospheric dispersal in the northern hemisphere quickly spread the burden to plants throughout all latitudes and to the southern hemisphere within another year or two. Figure 1 shows the chronological sequence of  $^{14}\text{C}$  addition as recorded in North American agricultural products. The initial input was a sharp pulse reflecting the rate of weapons testing, the moratorium of 1959-1960 and the very rapid rate of testing following resumption of tests by the USSR and US. This pulse of  $^{14}\text{C}$  has entered both marine and terrestrial food webs and can serve as a highly effective tracer of carbon movements on a multi-year scale. This poster illustrates three environmental studies using this tracer -- (1) identifying food sources for fishes and birds on Alaska's North Slope, (2) illustrating the importance of upwelling and the spring bloom to Bering Sea fauna and (3) as a tracer of plant productivity and decomposition during peat formation. Although these applications were to answer ecological questions, they also serve to illustrate pathways by which this pollutant has entered and been passed through the biosphere.

**Case 1. What is the role of peat carbon in the energetics of coastal arctic aquatic food webs?** The North Slope of Alaska is overlain by Holocene peats that have accumulated to depths of 0.5 - 2.0 m over much of the coastal plain. Peat carbon eroded from shorelines and transported by rivers supplies almost half of the carbon available to the nearshore (<10 km) arctic Alaskan coastal zone (Figure 2). To determine if this carbon was being utilized and passed up the food chains, the isotopic signatures of each potential source was compared with the  $^{14}\text{C}$  and  $\delta^{13}\text{C}$  values of fishes, birds, and prey organisms. Figure 3 shows these isotopic regimes and Figure 4 shows carbon isotope values of consumer organisms. Anadromous coregonids (whitefishes) that enter the marine environment in the summer to feed acquire isotopic signatures indistinguishable from obligate marine fishes, indicating a major dependence on marine food for their sustenance (Schell, 1983). Since peat carbon is a major fraction of the potential energy available to the invertebrate prey species in the lagoons where the fish feed, the data show that peat carbon, once in the marine environment, is not transferred to top consumers, but is probably lost in microbial respiration. This environment, because of shallow depths, small tides and a 1.5-2 m winter ice cover that freezes to the bottom in many areas, is depauperate of benthos. This may account for the very small fraction of terrestrial carbon evident in the fishes.

In freshwater, however,  $^{14}\text{C}$  depressions in fish show that a major fraction of their energy supply is peat-derived. Figure 5 shows radiocarbon content in grayling, *Thymallus arcticus*, which undergoes a major oscillation over the seasons. This indicates that the peat carbon, converted to biomass by insect larvae, is most important in winter months when cold and darkness keep photosynthesis at a minimum. Similar  $^{14}\text{C}$  depressions are evident in other species including the coregonids, although we do not have detailed seasonal data. Especially interesting are the ducks *Clangula hyemalis* which feed in the ponds on the tundra. Ponds too shallow for fish have high populations of herbivorous zooplankton;



whereas deeper ponds with fish have very few plankton and the ducks are forced to feed on benthic insects. The ducks feeding on zooplankton have high  $^{14}\text{C}$  content since phytoplankton production in the shallow ponds is closely coupling to atmospheric  $^{14}\text{CO}_2$  and the grazing zooplankton acquire the same  $^{14}\text{C}$  activity. Insectivorous ducks, however, have large  $^{14}\text{C}$  depressions reflecting mostly peat carbon in their prey. Figure 6 shows  $^{14}\text{C}$  activities in some tundra pond biota. In chironomids, the most abundant of the prey insects for fishes and many shorebirds, peat makes up over 50% of the body carbon. Other insect taxa also contain significant fractions of peat carbon in their body composition indicating alternate pathways of energy transfer to higher trophic levels. It is still unknown as to whether peat metabolism represents a means by which most modern pollutants are avoided in tundra food webs or if complexation of heavy metals and anthropogenic organic chemicals by peat serves to enhance assimilation into consumers via insect prey. These questions are acquiring increasing significance as oil industry development increases in the Arctic.

**Case 2. What is the significance of upwelling to the phytoplankton nutrition and food webs of the North Aleutian Shelf coastal waters and Bering Sea.** In a study of the nearshore zone of the North Aleutian Shelf, we found that the primary production of the region is heavily influenced by periodic upwelling onto the shelf and recycling of nutrients following the spring bloom (Schell and Saupe 1989). Figure 7 compares the  $^{14}\text{C}$  activities in fishes of the coastal zone with euphotic zone  $^{14}\text{C}$  activities in the inorganic carbon and macrophytes of the region. Also shown are the  $^{14}\text{C}$  depth profiles from the GEOSECS stations in the Bering Sea and the North Pacific (Ostlund and Stuiver, 1980). The fauna tend to have  $^{14}\text{C}$  activities toward the upper end of the scale indicating that most phytoplankton production occurs after there has been time to equilibrate carbon dioxide with the atmosphere. Although nitrate uptake associated with upwelling is important in the nutrient cycles as evidenced by some  $^{14}\text{C}$  depressed samples, recycled nutrients are most important for the bulk of carbon fixation in this shallow marine environment.

Bowhead whales give indication of similar processes of  $^{14}\text{C}$  equilibration into the oceanic euphotic zone and incorporation into the food webs in offshore waters of the Bering Sea. Radiocarbon activity in tissues taken from whales killed in 1981 are shown in Figure 8. Muscle, which is metabolically replaced in the order of 30 - 60 days, shows relatively uniform contamination by bomb  $^{14}\text{C}$ . This contrasts with wide variation in the  $^{14}\text{C}$  content of the subcutaneous blubber. Oil in insulative blubber, once formed, is apparently metabolically inactive, whereas lipids in internal fat and the innermost layer of blubber are metabolically active, being used by the whale in periods of fasting. Samples from large adult whales show  $^{14}\text{C}$  activity typical of pre-bomb levels indicating no replacement of blubber carbon. Younger whales which formed their blubber post-bomb have high  $^{14}\text{C}$  content (Schell et al., 1984). The long baleen plates in the mouths of bowhead whales take up to 20+ years to grow. Radiocarbon activity measured along their lengths shows the sharp increase in  $^{14}\text{C}$  activity during the period 1962-5 as the atmospheric  $^{14}\text{C}$  burden equilibrated with the ocean surface waters (Schell et al., 1989).

**Case 3** Can the temporal changes in bomb radiocarbon activities in vegetation be used to give indication of decomposition rates of tundra vegetation and the current rates of peat accumulation. The possible effects of global warming on primary productivity in tundra and decomposition are critical in trying to establish the role of arctic peatlands in the global carbon cycle. Tundra is a major store of subfossil carbon and some investigators claim recent changes in climate have caused drying and increased respiration leading to an excess of respiration over fixation. Northern hemisphere peatlands are so large that an increase in the acrotelm (oxic zone) of only 10 cm would lead to an increase in respiration and CO<sub>2</sub> production that would be second only to fossil fuel combustion.

Figure 9 shows the profiles of <sup>14</sup>C in the moss-peat layer at Imnavait Creek, north of the Brooks Range in arctic Alaska. Also shown is a profile of <sup>137</sup>Cs (from Grebmeir et al., in press) collected from the same location. The close similarities between the two profiles are indicative of the accumulation of bomb radionuclides and little post-depositional leaching. Since the <sup>14</sup>C activities in the vegetation closely reflect the atmospheric levels at the time of carbon fixation, the year of deposition can be assigned to each subsample taken from the moss layer. Immediately beneath the moss, the <sup>14</sup>C content drops sharply and represents the accumulation of peat. Figure 10 shows a log plot of the carbon content of each sample -- the intercept on the Y-axis is the primary production that fell as litter and the intercept at the permafrost horizon represents the rate of peat accumulation, about 2.8 g C/m<sup>2</sup>-yr. No indication of significant change in accumulation or decomposition rates are evident over the depth of the core suggesting modern rates of primary productivity, decomposition, and peat formation are similar to those in the recent past.

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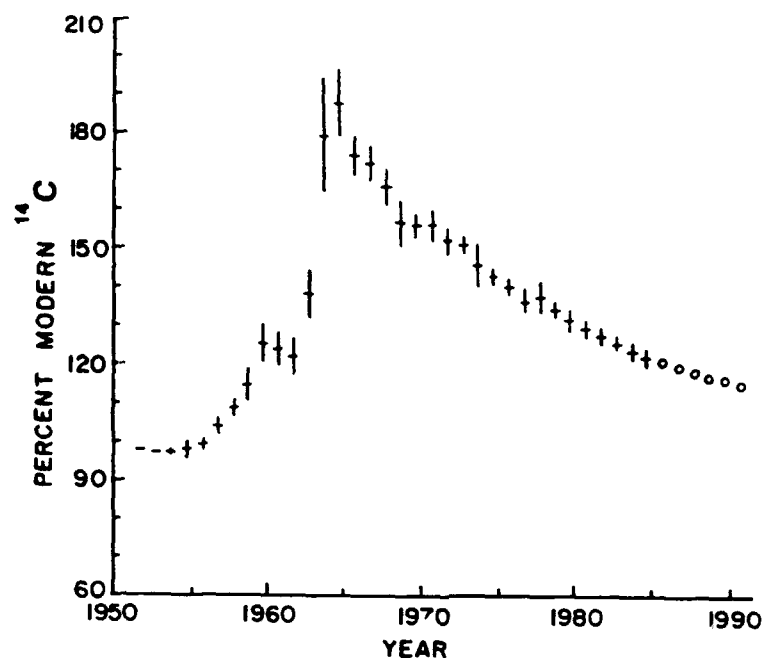


Fig. 1. Average annual  $^{14}\text{C}$  activities in North American agricultural products.

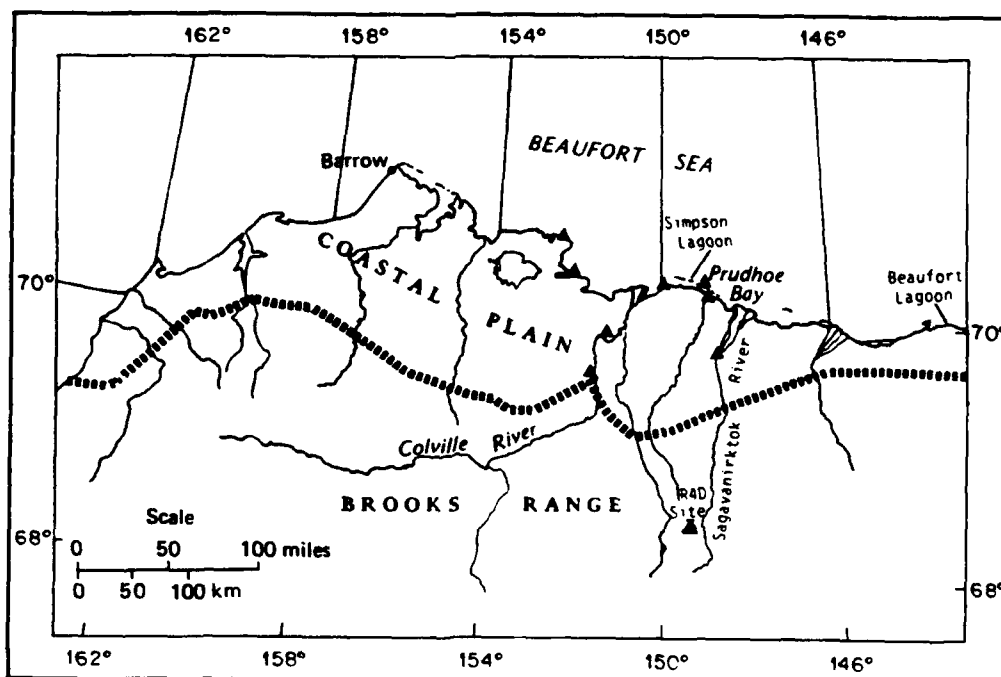


Fig. 2. Arctic Alaska. The food chain studies were conducted in the ponds, rivers and lagoons in the vicinity of Simpson lagoon. The research on tundra carbon accumulation was conducted at Imnavait Creek at the R4D research site on the North Slope of the Brooks Range. The bowhead whale study was conducted on whales harvested by natives at Point Barrow. The whales migrate between the Bering Sea in winter to the eastern Beaufort Sea in summer.

Separation of  
Peat and  
Modern carbon

Separation of  
Marine and  
Terrestrial carbon

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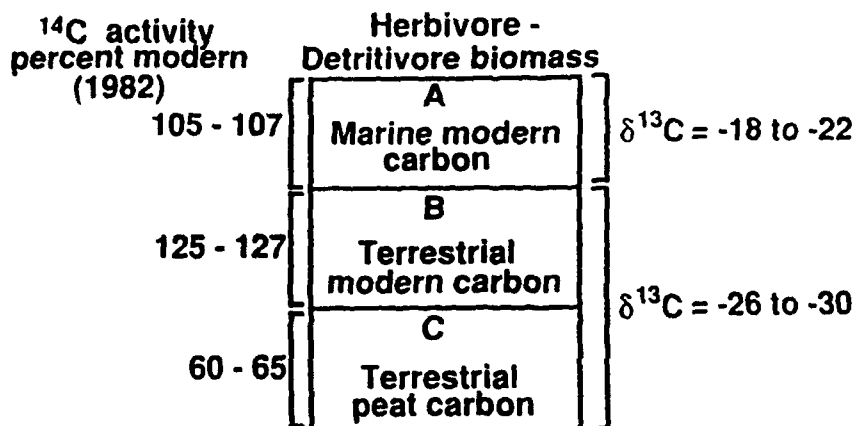


Figure 3. Carbon isotope domains (1982) used as end members in these studies.

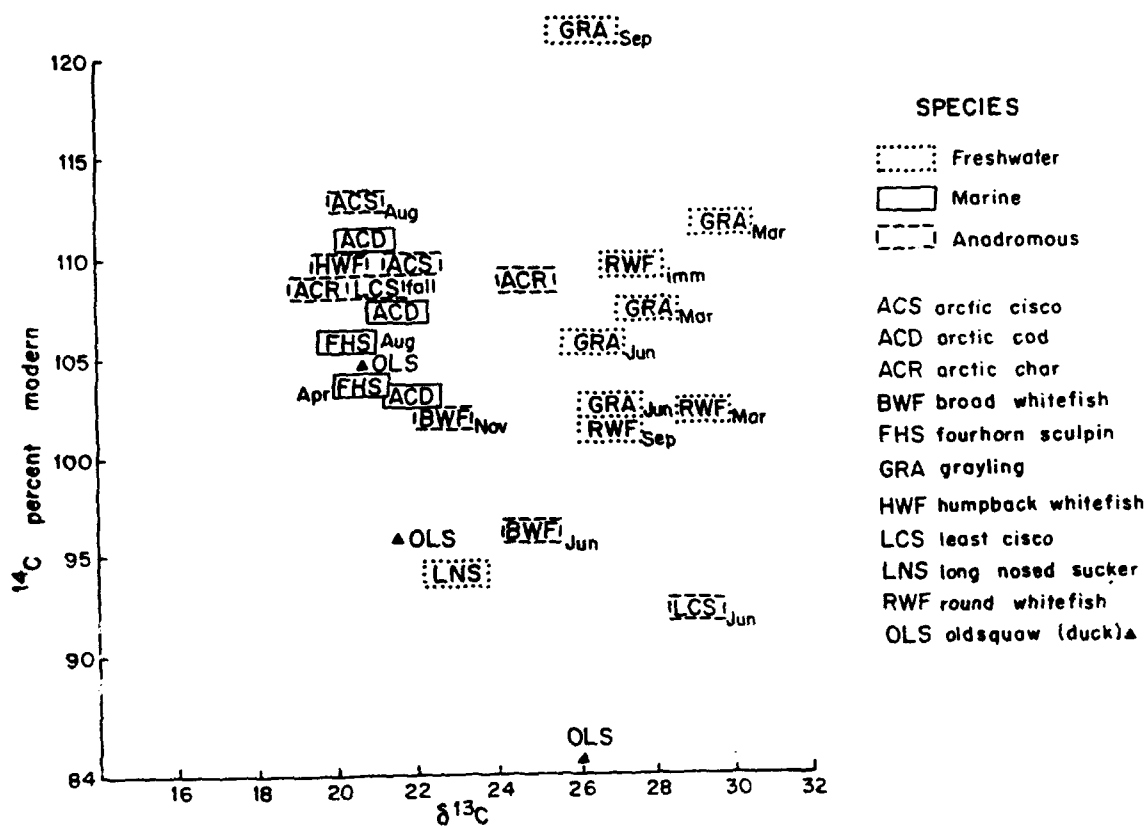


Figure 4. Stable and radiocarbon isotope data on fauna from the Alaskan North Slope.

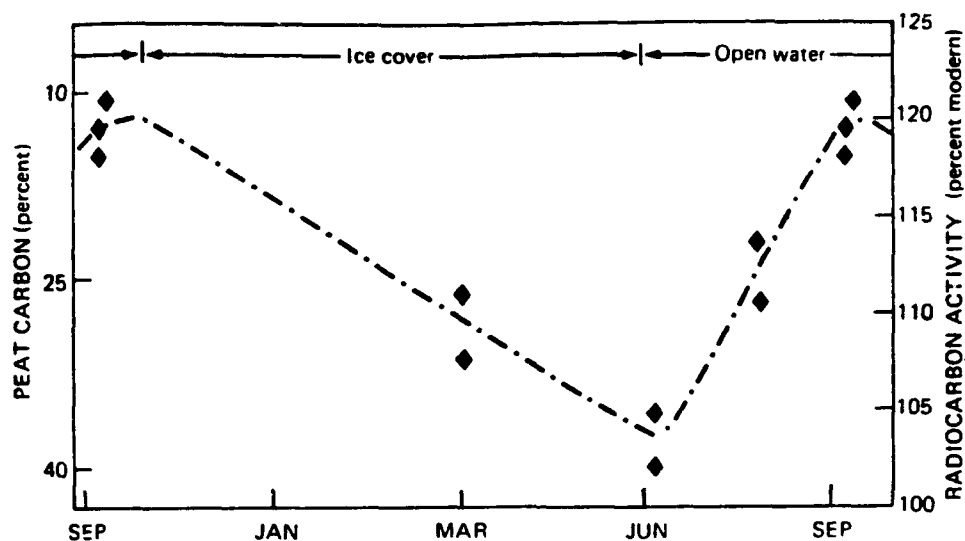


Figure 5. Seasonal radiocarbon data for the fish *Thymallus arcticus* (grayling). Much of the annual cycle is beneath ice cover in darkness during which time peat-based food webs provide insect prey.

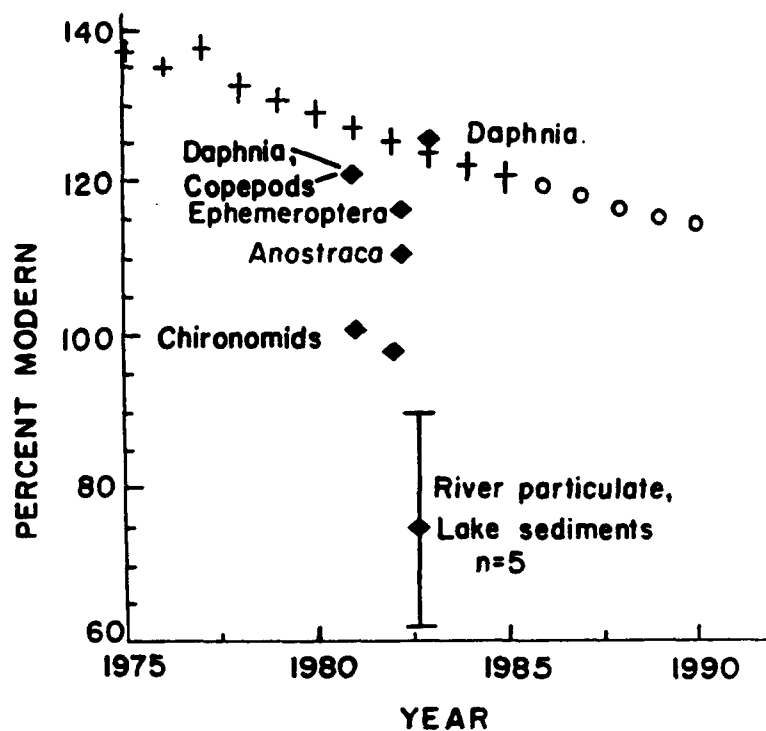


Figure 6. Invertebrate prey used by fish in tundra ponds and lakes. The crustaceans *Daphnia* spp. are filter feeders and dependent upon modern primary production. *Anostraca* and insects show varying amounts of peat carbon composition evidenced by  $^{14}\text{C}$  activity depressions. The sediments are lake bottom surface sediments and suspended particulate matter in the Colville River. Crosses represent atmospheric  $^{14}\text{C}$  activity prior to sample collection, circles were predicted values.

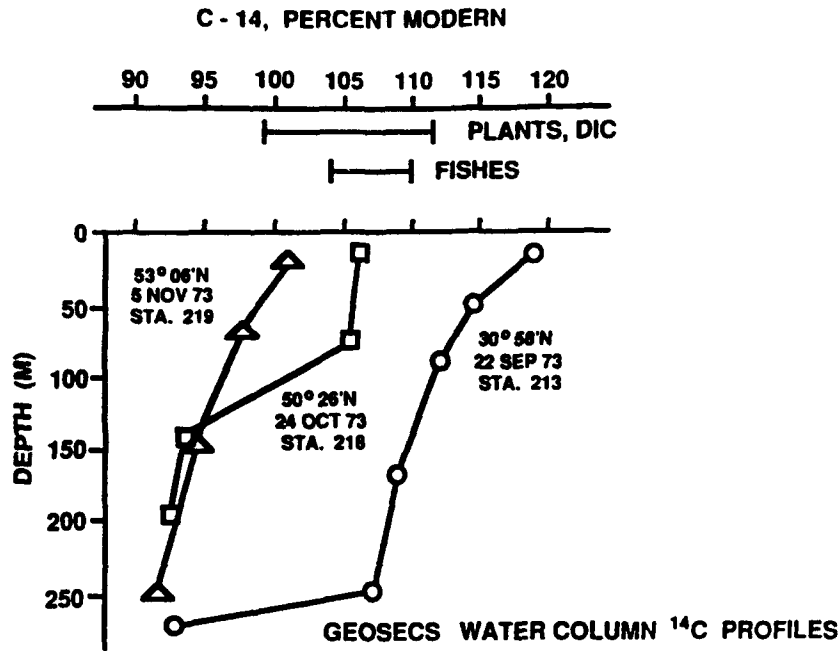


Figure 7. Radiocarbon activities in the biota and environment of the North Aleutian shelf coastal zone. Vertical profiles of inorganic carbon activities are from GEOSECS data (Ostlund and Stuiver, 1980).

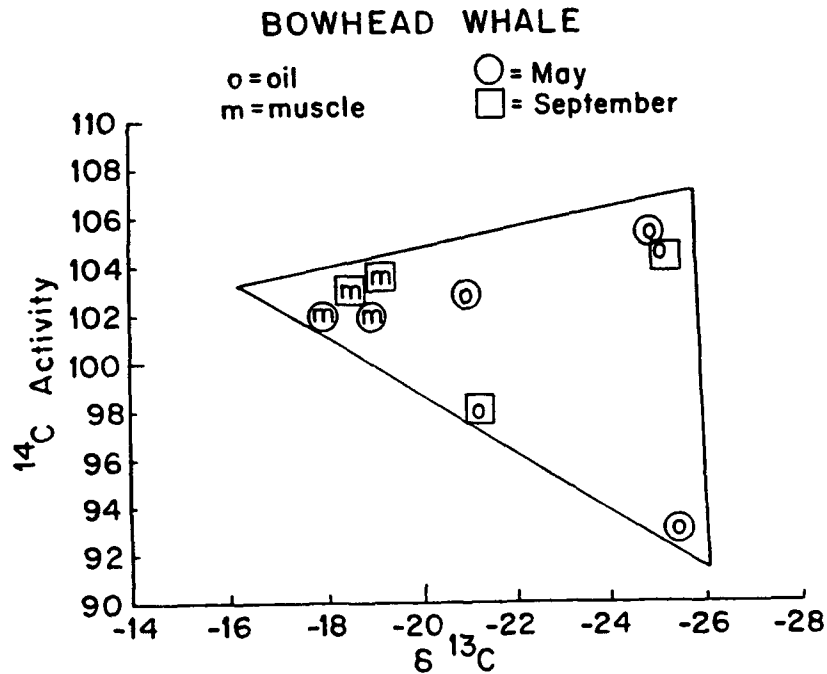


Figure 8. Bowhead whale  $^{14}\text{C}$  activities in muscle and blubber. Muscle carbon turns over rapidly and all whales show similar values whereas subcutaneous blubber oil is metabolically inactive following deposition and reflects the  $^{14}\text{C}$  activity of the years of formation.

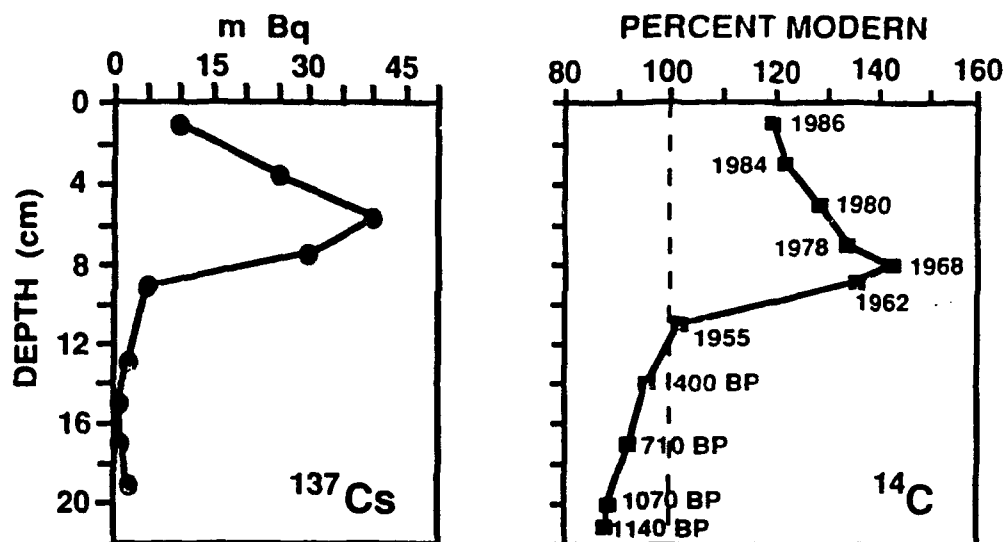


Figure 9. Radiocarbon profile through the moss layer and underlying peat at Imnavait Creek, Alaska. Years were assigned based upon data in Figure 1. The  $^{137}\text{Cs}$  data (Grebmeir et al., in press) closely match the  $^{14}\text{C}$  curve suggesting little post-depositional leaching.

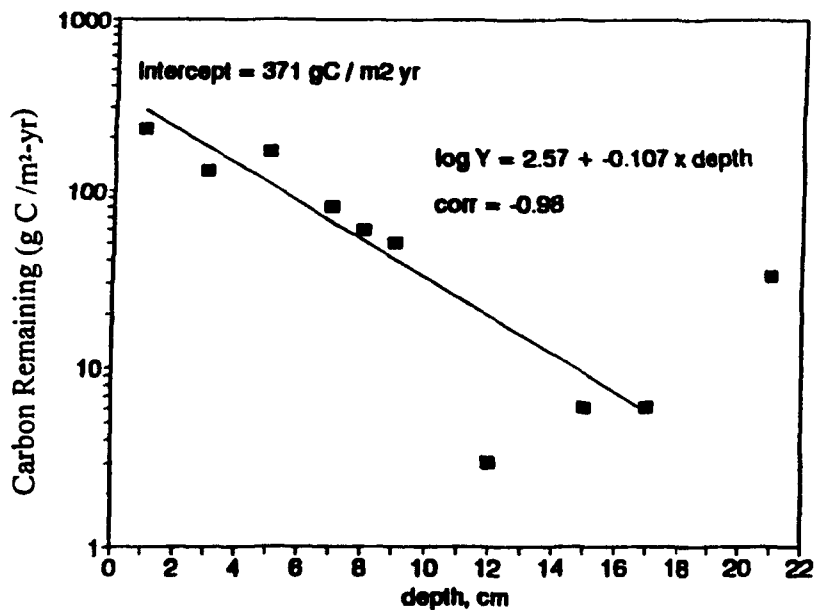


Figure 10. Litter decomposition at Imnavait Creek. Carbon remaining at the year indicated from an initial primary production of  $371 \text{ g C} / \text{m}^2\text{-yr}$ . The permafrost horizon is approximately 20 cm depth, below which no further decomposition is presumed to occur.



**$^{129}\text{I}$  as a tracer of European reprocessing emissions  
in the North Atlantic and Arctic Oceans**

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**Abstract :** In the pre-nuclear era, the concentration of  $^{129}\text{I}$  (half-life 16 My) in the ocean was  $\sim 3 \times 10^5$  atoms/l. The input of this isotope from nuclear weapons testing,  $\sim 2 \times 10^{26}$  atoms, increased this concentration in ocean surface water to  $\sim 3 \times 10^7$  atoms/l. On the basis of measurements we have made on archived seaweed samples, together with available release data, we tentatively estimate that the input of this isotope to the oceans from the nuclear fuel reprocessing facilities at La Hague, France, and Sellafield, Great Britain, during the past 25 years has been  $\sim 5 \times 10^{27}$  atoms, or  $\sim 1.2$  ton of  $^{129}\text{I}$ . Most of this  $^{129}\text{I}$  is transported up the west European coastline, and into the North Atlantic and Arctic oceans. The technique of accelerator mass spectrometry (AMS) is capable of measuring  $10^6$  atoms of  $^{129}\text{I}$ , thus offering an extremely sensitive method of tracing this isotope in the oceans. We have begun such studies, and will show, for example, that one can detect the reprocessing signal in 1 litre seawater samples from virtually anywhere in the North Atlantic. We also discuss the potential of using this isotope to trace other possible intentional or accidental releases of fission products in the oceans. Because a ratio such as  $^{129}\text{I}/^{137}\text{Cs}$  from the nuclear reprocessing plants is much larger than that from weapons fallout, Chernobyl, or other unprocessed waste, it can provide a very effective "fingerprint" for distinguishing between these different potential sources of observed radioactivity in the oceans.

## INTRODUCTION

$^{129}\text{I}$  is a radioactive isotope with a half-life of 16 My. Before the nuclear era, when  $^{129}\text{I}$  on the earth was principally due to spontaneous fission of  $^{238}\text{U}$  and cosmic ray induced spallation of Xe in the atmosphere, the ratio of  $^{129}\text{I}/\text{I}$  in the ocean is estimated to have been  $\sim 10^{-12}$  (Fabryka-Martin et al., 1985 ; Fehn et al., 1986 ; Baba, 1990 ; Kilius et al., 1992).

In the post nuclear era, anthropogenically produced  $^{129}\text{I}$ , principally from nuclear reactor operation and nuclear bomb testing, overwhelmingly dominates natural production. The total release of  $^{129}\text{I}$  in nuclear weapons testing is estimated to have been  $\sim 2 \times 10^{26}$  atoms (Chamberlain, 1991). Assuming this quantity has been homogenized in an ocean mixed surface layer of 100 m depth, and using an ocean  $^{127}\text{I}$  concentration of 60  $\mu\text{g/l}$ , (Wong, 1991) the resulting  $^{129}\text{I}/^{127}\text{I}$  ratio would be  $\sim 2 \times 10^{-11}$ . This is within the range, but somewhat lower than the average ( $\sim 10^{-10}$ ) of the relatively few available measurements on post bomb marine material (Fehn et al., 1986 ; Baba, 1990 ; Kilius et al., 1992). This may be due to incorporation into some of those samples before complete homogenization had occurred.

In the thermal neutron induced fission of  $^{235}\text{U}$ ,  $^{129}\text{I}$  is produced with a yield of  $\sim 0.6\%$ . During the reprocessing of spent nuclear fuel, a fraction of the  $^{129}\text{I}$  so formed is released to the environment. The largest nuclear fuel reprocessing facilities presently operating are located at La Hague, France and Sellafield, Great Britain.

About 3 years ago, we recognized that this reprocessing derived  $^{129}\text{I}$  was potentially a very powerful oceanic tracer. Although it is biophilic (and thus may also be useful as a "marker" of organic matter formation, transport and regeneration) the observed residence time of iodine in the ocean ( $\sim 10^5$  years) suggests that it behaves almost conservatively in the open sea. An advantage of  $^{129}\text{I}$ , compared to certain other transient tracers (chlorofluorocarbons, nuclear bomb derived fallout,  $^{85}\text{Kr}$ ) is its more specific spatial source function. Thus its use, either alone or in combination with one of the above tracers (Livingston et al., 1985 ; Schlosser et al., 1991), might help better understand the formation of North Atlantic deep water, which is a subject of continuing interest (Dickson et al., 1990 ; Stocker and Broecker, 1992). The advantages of  $^{129}\text{I}$  compared to other reprocessing derived nuclides are, as shown below, the much smaller sample required for analysis, and a reduced interference from a weapons or Chernobyl produced component.

Because of its long half-life, determination of  $^{129}\text{I}$  by radioactive decay counting is quite inefficient. Several other techniques, including activation analysis, low energy mass spectrometry and laser spectroscopy have been used (for a brief review, see Filistovich et al., 1986) but have not found widespread adoption. This is presumably due to the rather sophisticated instrumentation necessary and, perhaps even more important, the fact that the sensitivity of these alternative techniques is generally limited by the value of the  $^{129}\text{I}/^{127}\text{I}$  ratio in the sample to be measured. AMS was first applied to the determination of  $^{129}\text{I}$  more than 10 years ago, using a very large tandem accelerator (Elmore et al., 1980). Kilius et al., (1990) have recently demonstrated that, by using sufficiently high resolution electrostatic and electromagnetic analysis both before and after

the accelerator, a small tandem accelerator can give AMS measurements of comparable quality to the larger systems.

We thus decided to use the IsoTrace AMS facility to investigate the potential of reprocessing derived  $^{129}\text{I}$  as an oceanographic tracer. We first present here our estimates of the marine input functions of  $^{129}\text{I}$  from La Hague and Sellafield. We then briefly summarize our initial investigations on the transport of this  $^{129}\text{I}$  into the North Atlantic. (Raisbeck et al., 1992). Finally, for the purpose of the present meeting, we discuss how  $^{129}\text{I}$  can be used to distinguish between radioactivity in the oceans coming from the European reprocessing facilities, and that coming from other deliberate or accidental releases of unprocessed fission products.

## INPUT FUNCTIONS

In order to most fully exploit the potential of  $^{129}\text{I}$  as an ocean tracer, it is necessary to have reliable estimates of the temporal input function of this isotope. For the moment, complete records of the reprocessing emissions are unavailable and/or unpublished. In order to obtain such information, we have measured  $^{129}\text{I}/\text{I}$  in archived seaweed samples collected from near the reprocessing discharge points. We then compare, and where possible normalize, these results to the available emission data in order to make preliminary estimates of the temporal input functions from these two plants.

## EXPERIMENTAL PROCEDURE

Samples (1 per year) of dried archived seaweed were obtained from Goury, France and Seascale, U.K. The sampling point at Goury is located ~ 5 km north of the discharge pipe from the La Hague reprocessing plant. Previous studies have shown that a significant fraction of the emissions have a transit time of several months from the discharge point to the location where the seaweed (*fucus serratus*) was collected (Fraizier et al., 1992).

The Seascale samples (porphyra) were taken ~ 3 km south of the Sellafield discharge point. As we shall see below, this proximity is probably a disadvantage for the present application.

Red and brown seaweeds are known to have significant concentrations (~ 50-500 ppm) of iodine. The simplest way of determining  $^{129}\text{I}/\text{I}$  in such samples is thus to measure the ratio of extracted iodine directly. In the case of the samples considered here, these ratios would, as will be seen below, be very high ( $> 10^{-6}$ ). However, the accelerator mass spectrometer (AMS) facility that we are using is very sensitive, and is used to measure  $^{129}\text{I}/\text{I}$  ratios as low as  $10^{-14}$ . Thus a direct measure of extracted iodine would risk introducing an unacceptably high level of  $^{129}\text{I}$  into the accelerator ion source, thus possibly compromising the background level for other measurements. To avoid this, the procedure adopted here was one of isotope dilution.

The dried seaweed sample (20 - 300 mg) was first digested by hot concentrated NaOH in a quartz crucible. The dried residual was then ashed in a muffle furnace at 650°C for 1 hour. The iodine was leached from the ash with hot water. An aliquot of this leach solution was reserved for stable iodine analysis by a colorimetric procedure, adapted from Barkley and Thompson, (1960). To the remaining leach solution was added 6 mg of stable iodine carrier, as KI. The iodine was

transformed to  $\text{IO}_3^-$  to ensure isotopic equilibrium, reduced to  $\text{I}_2$  and extracted into  $\text{CCl}_4$ , further reduced to  $\text{I}^-$  and backextracted into aqueous solution, and finally precipitated as  $\text{AgI}$ . The dried  $\text{AgI}$  was mixed with niobium powder, and the  $^{129}\text{I}/\text{I}$  ratio measured at the IsoTrace AMS facility (Kilius et al., 1990). From this ratio, and the stable iodine measurement, one can calculate the  $^{129}\text{I}/\text{I}$  in the original seaweed.

## RESULTS

Our results are shown in Figs. 1 and 2. Also shown in these figures are all the official  $^{129}\text{I}$  discharge data we have been able to obtain.

### *La Hague*

The earliest seaweed we were able to obtain was from 1980. As can be seen in Fig. 1, the  $^{129}\text{I}/\text{I}$  ratio in the seaweed from Goury has been relatively constant from 1980 - 1990, in good agreement with the discharge data. Beginning in 1990 when a new plant was put into operation at La Hague, both discharge and measured ratios increased by a factor of  $\sim 3$ . It therefore appears the  $^{129}\text{I}/\text{I}$  ratio in Goury seaweed is a fairly reliable "proxy" for  $^{129}\text{I}$  discharge levels from La Hague. This is particularly gratifying to us since, when estimates based on these seaweed data were first presented publicly, the only discharge data available to us was for 1983-1986 (Patti et al., 1988), and some skepticism was expressed regarding the reliability of our procedure.

To quantify the above relationship, we define a parameter  $D/R$ , where  $D$  is the given discharge, in kg, for the  $\sim 12$  months prior to the sampling date (Feb) and  $R$  the measured  $^{129}\text{I}/\text{I}$  seaweed ratio, in units of  $10^{-6}$ . For the period 1983 - 1992 this parameter is equal to 21.7, with a standard deviation ( $\sigma$ ) of 25 % and standard deviation of the mean ( $\sigma_m$ ) of 7.8 %. Using this same relationship for the period 1979 - 1982, we obtain the estimates shown as open squares in Fig. 1b, and discharges for that period of  $94 \pm 12$  kg.

Also shown in Fig. 1 is the quantity of fuel reprocessed from 1975 - 1992 (J. Pijlselman, private communication). One can see that the  $^{129}\text{I}$  discharge correlates quite well with this quantity. As above, we can define a parameter  $D/T$ , where  $T$  is the quantity of fuel reprocessed, in tons. For the period 1983 - 1992, this parameter is equal to  $83.1 \times 10^{-3}$ , with  $\sigma$  of 27 % and  $\sigma_m$  of 8.4 %. It is interesting to note in passing that, based on the typical concentration of  $^{129}\text{I}$  in spent fuel rods ( $\sim 170$  g/ton) the fraction of this isotope released to the oceans from the reprocessing treatment is  $\sim 50$  %. Assuming this relationship has been the same in the period prior to 1980, we can make an estimate ( $121 \pm 16$  kg) for the  $^{129}\text{I}$  discharge for the period 1975 - 1979 (stars in Fig. 1b).

Using these estimates, for periods where no official release data exist, the total  $^{129}\text{I}$  discharge from La Hague from 1975 - 1992 is found to be  $\sim 632$  kg.

It is worthwhile to note that the sudden increase in  $^{129}\text{I}$  emissions from La Hague in 1990 may be useful as a time marker for tracing these emissions along the European coastline and into the North Atlantic and Arctic oceans.

## Sellafield

For Sellafield we have seaweed samples from 1967 - 1991 (except 1976). As can be seen in Fig. 2a, the measured  $^{129}\text{I}/\text{I}$  ratio is both larger and more irregular than at Goury. This is undoubtedly due to the location of the sampling point relative to the discharge pipe. The results thus appear to be reflecting rapid variations in the  $^{129}\text{I}$  discharges, which lead to rapid variations in the  $^{129}\text{I}/\text{I}$  in the nearby seawater. While one can expect seaweed to "integrate" the seawater signal of radioactive species over a certain time (Dahlgaard and Boelskifte, 1992), we have no quantitative information on this for iodine in porphyra.

Probably because of the sensitivity of the seaweed  $^{129}\text{I}/\text{I}$  ratio to rapid variations in seawater values, the measured time profile does not track very well the available yearly discharge data (Fig. 2b). Since our samples were all from approximately the same period of the year (May or June), this suggests a different monthly discharge pattern, from one year to another (this is in fact confirmed for those years where monthly or quarterly discharge data is available).

If, as before, we consider the parameter  $D/R$ , for the period 1978 - 1991 at Sellafield, we obtain a value of 4.55, with  $\sigma$  of 58% and  $\sigma_m$  of 16 %. Using this relationship for the period 1967 - 1977, we obtain the open squares in Fig. 2b, and total iodine emissions for that period of  $314 \pm 57$  kg. Combining the above estimate with the official release figures gives total  $^{129}\text{I}$  discharges from Sellafield from 1967 - 1992 as 608 kg.

For Sellafield, we do not have information on the quantity of fuel reprocessed annually. However, given that this facility processes both military and civilian fuels, which have quite different burnup parameters, it is in any event not obvious that the  $^{129}\text{I}$  discharges could be reliably predicted from the quantity of fuel treated.

One way of improving the data from Sellafield would be to use seaweed taken further away from the discharge point, so that it is sampling a more smoothed emission signal. Appropriate samples are apparently available (P. Kershaw, private communication), and we are presently planning such measurements.

In summary then, while refinements can undoubtedly be made, the data described above suggest that the combined total  $^{129}\text{I}$  discharge to the oceans by the reprocessing plants at La Hague and Sellafield has been  $\sim 1.2$  tons, or  $\sim 5 \times 10^{27}$  atoms. This is an order of magnitude larger than the total estimated  $^{129}\text{I}$  in the pre-nuclear era ocean, approximately 25 times the input due to nuclear weapons testing, and several hundred times that released by Chernobyl. It is expected therefore that the reprocessing signal will dominate these other sources of  $^{129}\text{I}$  in the North Atlantic and Arctic oceans.

## TRANSPORT INTO THE NORTH ATLANTIC AND ARCTIC OCEANS

In order to indicate its dispersion in the North Atlantic we give in Table 1 measurements of  $^{129}\text{I}$  in samples of seaweed and seawater taken at various locations as shown in Fig. 3. (Raisbeck et al. 1992).

Using the results given in Table 1, one can estimate the minimum sample requirements for  $^{129}\text{I}$  analyses in the region studied. Thus in the English Channel or Irish Sea, such determinations could be made on water of less than 1 ml (although there is no apparent advantage in working with such small quantities) while 1 litre samples appear sufficient for anywhere in the North Atlantic. This can be compared to other reprocessing derived nuclides ( $^{137}\text{Cs}$ ,  $^{90}\text{Sr}$ ,  $^{99}\text{Tc}$ ) that often require in this region tens or even hundreds of litres for analysis.

Knowing their stable iodine content, one can also estimate the minimum sample size for various other potential marine samples (plankton, shellfish, sediment, etc). One finds, for example, that over the whole area studied, one could measure  $^{129}\text{I}$  in a single oyster !

### **$^{129}\text{I}$ AS A TRACER OF REPROCESSING EMISSIONS**

As should be evident from the preceeding sections, the fraction of  $^{129}\text{I}$  discharged to the oceans by the reprocessing facilities at Sellafield and La Hague is much larger than for most other fission products. Therefore  $^{129}\text{I}$  can be used as a very sensitive and specific tracer of the reprocessing signal, even very far from its origin. For example the atom ratio  $^{129}\text{I}/^{137}\text{Cs}$  from the above cited reprocessing facilities is currently  $\geq 10^3$ , while the same ratio from unprocessed fission products is  $\sim 0.1$ . Thus by measuring  $^{129}\text{I}/\text{X}$  (where X is another soluble nuclide such as  $^{137}\text{Cs}$ ,  $^{90}\text{Sr}$ ,  $^{99}\text{Tc}$  etc) in an ocean sample, one can estimate what fraction of X originated from reprocessing and, by implication, what fraction came from other sources. Such a possibility should be very useful for trying to identify a signal from accidental or deliberate discharges of unprocessed waste. To this end, we are presently carrying out  $^{129}\text{I}$  measurements in water from in and around the Kara Sea, and the first results will be reported at the Kirkenes meeting, in Aug 1993.

Naturally, if reprocessing waste from Russia or the former Soviet Union have been directly or indirectly (via rivers) released into the Arctic oceans, it is quite possible that they also were enhanced in  $^{129}\text{I}$  relative to other fission products. It is therefore important that as much information as possible on the quantity and composition of such releases be made available. In addition to measurements in present river water, one possible way of obtaining information on past releases would be to analyse sediment cores from the relevant river estuaries.

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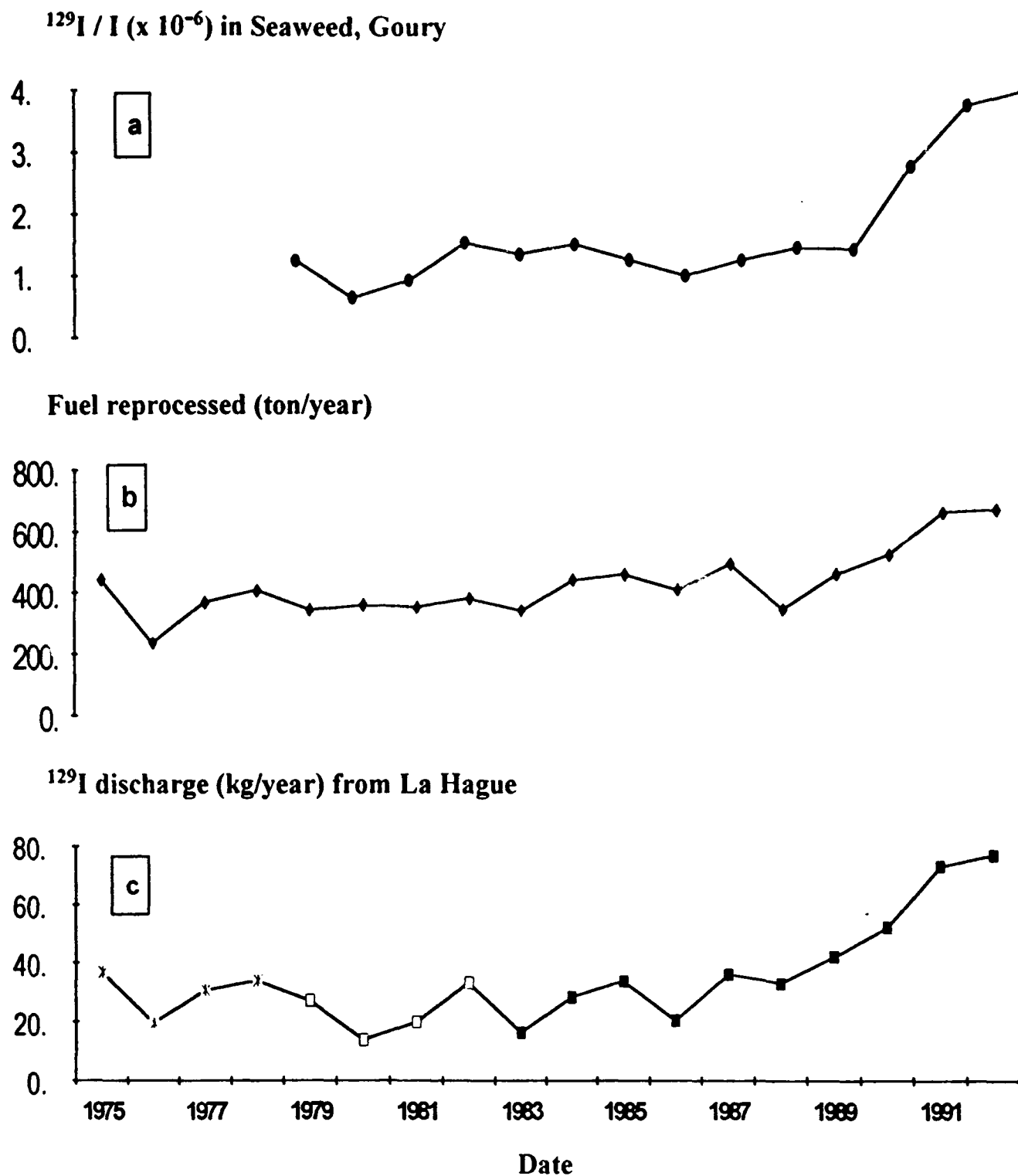


Table 1  
Sample Characteristics and Experimental Results

Location	Map code	Sample date	Sample type	Water depth (m)	Sample wgt(g) or vol (ml)	$^{129}\text{I}/^{127}\text{I}$ measured ( $10^{-11}$ )	$^{127}\text{I}$ (ppm)* or ( $\mu\text{g/l}$ )	$^{129}\text{I}/^{127}\text{I}$ in sample ( $10^{-10}$ )
Doélan (a)*	1	1/9/91	seaweed	0	17.53	95±20	-	9.5±2.0
" (b)*		"	"	0	19.27	97±20	-	9.7±2.0
" (a)		1/12/91	water	0	100	0.29±0.04	120±6.0	12.3±1.8
" (a)		1/3/92	"	0	100	0.13±0.02	52.1±2.6	12.7±2.1
Roscoff	2	26/6/85	seaweed	0	11.76	50.3±1.1	990±50	500±30
Cancale	3	9/6/91	"	0	11.30	343±23	300±15	3400±300
Carteret	4	4/2/91	"	0	18.42	124±2	650±65	7600±800
Herquemoulin	5	15/6/84	"		12.41	108±3	525±53	10000±1000
		6/2/91	"	0	11.03	213±6	690±70	37000±3800
Fermanville	6	5/2/91	"	0	12.07	185±5	754±75	15000±1500
Gatteville	7	5/4/91	"	0	7.40	796±63	310±15	8000±700
Luc sur Mer	8	15/1/91	"	0	11.18	310±19	340±17	3100±250
Honfleur	9	27/2/91	"	0	10.75	445±8	450±19	4500±240
49°52'N, 00°00'E	10	1/12/91	water	0	100	27±3	55.5±2.8	4900±600
Gravelines	11	11/1/91	seaweed	0	11.45	311±7	790±40	3100±200
Lowestoft	12	20/3/92	water	0	100	3.9±0.5	46.5±2.3	430±60
Heysham	13	5/4/92	"	0	100	32±4	36.5±1.8	8900±1200
Maryport	14	5/4/92	water	0	100	18±2	36.2±1.8	5100±600
56°00'N, 06°00'E	15	27/11/91	"	0	100	2.4±0.3	52.8±2.6	460±60
Lossimouth	16	7/4/92	"	0	100	1.8±0.2	57.8±2.9	160±20
61°20'N, 7°53'W	17	20/8/90	"	200	900	0.32±0.04	58.0±2.9	3.1±0.4
Thorshavn	18	18/7/89	seaweed	0	17.31	29±4	-	2.9±0.4
Sørvagur	19	15/7/89	"	0	10.01	39±5	-	3.9±0.5
63°50'N, 6°05'W	20	19/8/90	water	50	300	0.29±0.03	52.9±2.7	9.3±1.1
"		"	"	200	300	0.23±0.04	52.8±2.6	7.4±1.3
"		"	"	1000	300	0.29±0.03	58.9±3.0	8.4±1.0
Vestmarnaeyjar	21	6/12/91	seaweed	0	18.01	24±5	-	2.4±0.5
Grimsy	22	15/12/90	"	0	10.00	29±6	-	2.9±0.6
		6/12/91	"	0	17.60	39±6	-	3.9±0.6
Raudfjorden	23	8/9/92	"	0	1.94	26.0±0.5	194±8	37.2±1.2
Adventfjorden	24	8/9/92	"	0	1.92	0.23±0.004	1.90±6	32.2±1.1

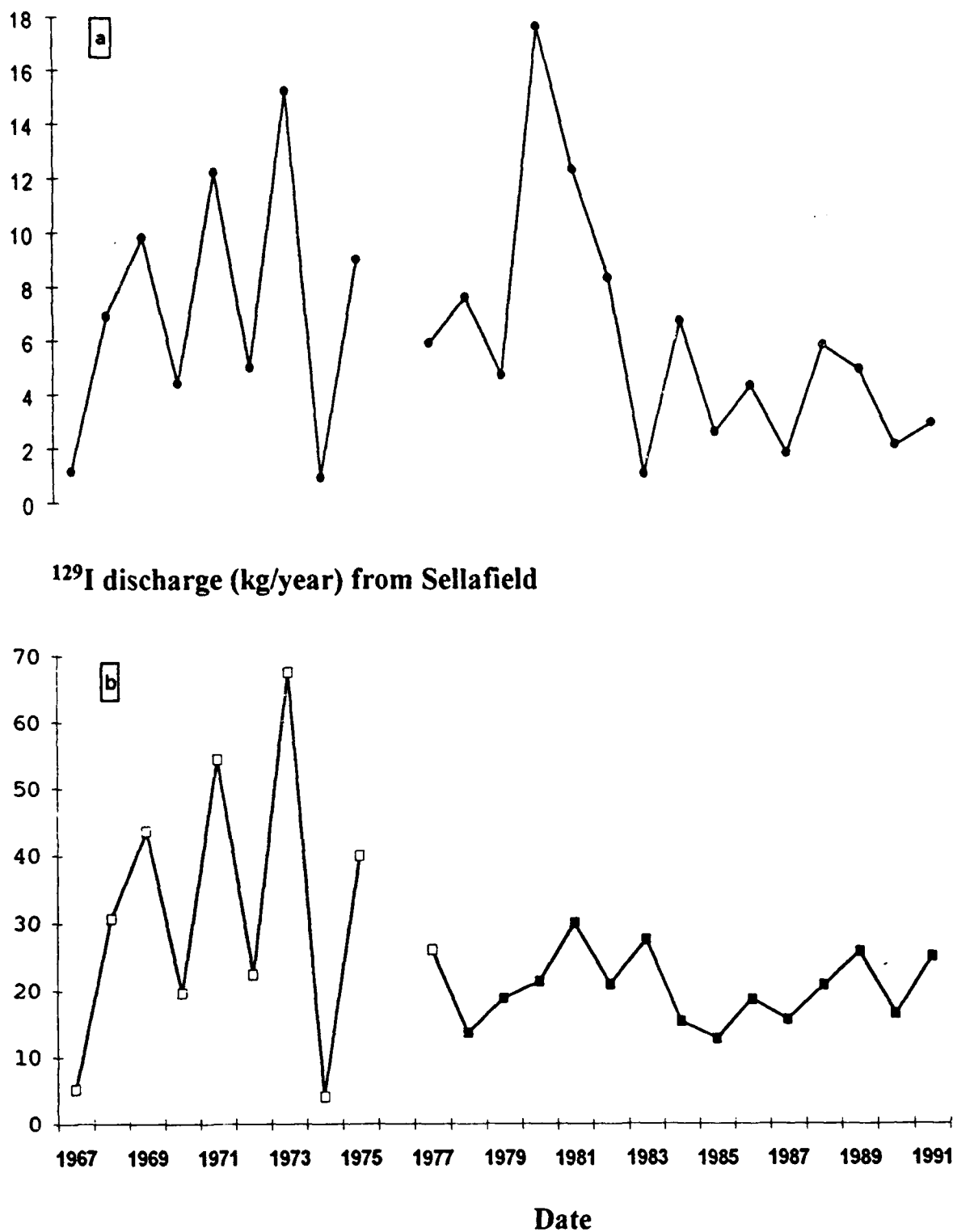
\* Locations Doélan (a) and (b) are less than 2 km apart

\* For seaweed this concentration is the amount extracted (expressed as ppm of dry seaweed wgt). This is the relevant factor for determining dilution factor for  $^{129}\text{I}$ , even though it may not represent total  $^{127}\text{I}$  in seaweed.



**Figure 1 :** (a)  $^{129}\text{I} / \text{I}$  in seaweed from Goury, (b) quantity of fuel reprocessed at La Hague, and (c) marine discharges of  $^{129}\text{I}$  from La Hague based on official release data (Radioprotection 23, 381-392 (1988) and J. Pijselman (private communication)) (■), or estimated from seaweed measurements (□) or quantity of fuel reprocessed (\*).

# $^{129}\text{I} / \text{I} (\times 10^{-6})$ in Seaweed, Seascale



**Figure 2 :** (a)  $^{129}\text{I} / \text{I}$  in seaweed at Seascale, (b) marine discharges of  $^{129}\text{I}$  from Sellafield based on official release data (BNFL annual reports and D. Woodhead, (private communication)) (■), or estimated from seaweed measurements (□).

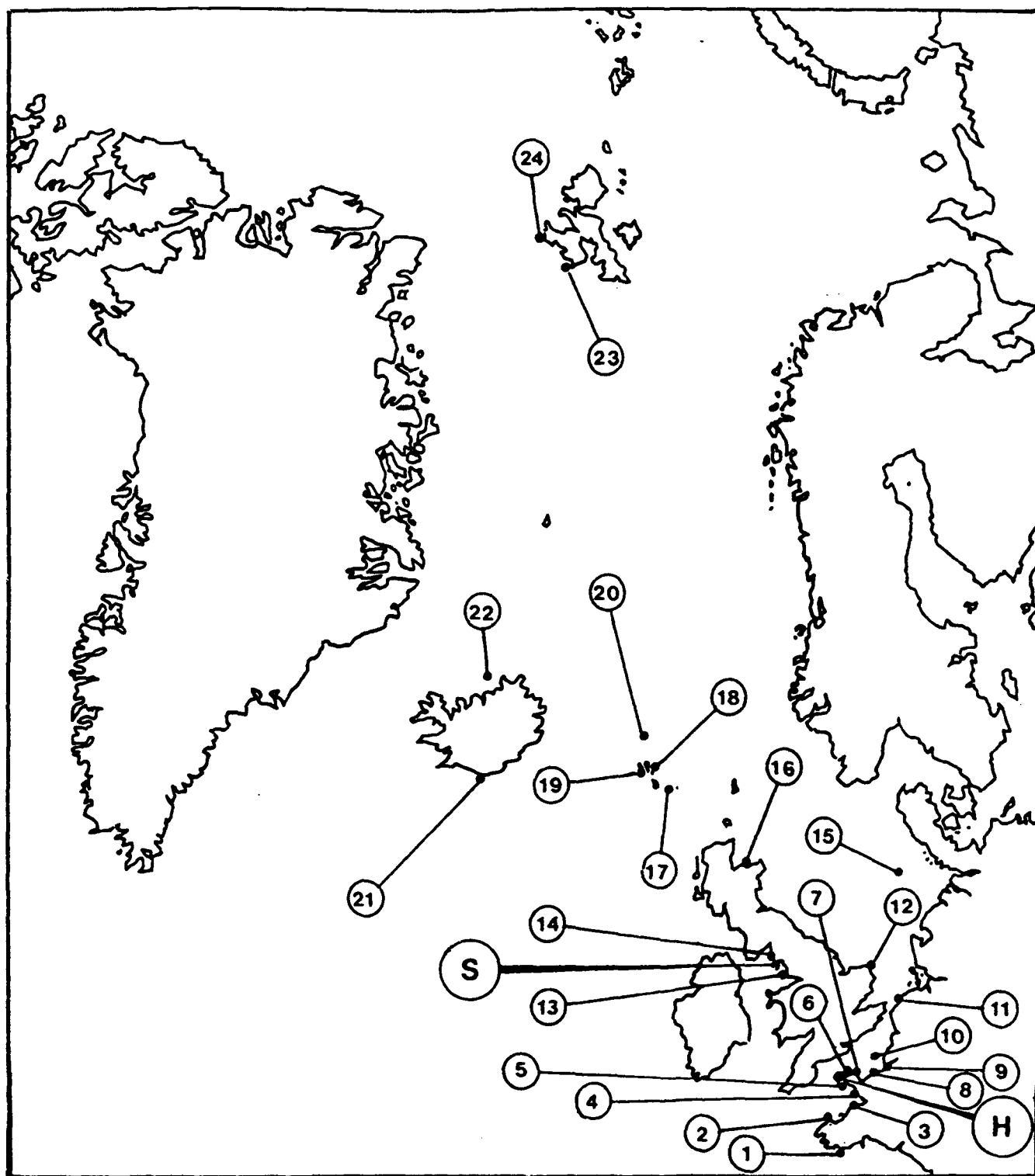


Fig. 3: Map showing sample locations. H and S indicate the reprocessing plants at La Hague and Sellafield, respectively.

## Radioactive Contamination of the Barents and Kara Seas

S. Vakulovskiy, A. Nikitin, V. Chumichev

The monitoring of radioactive contamination of Arctic seas by the Russian hydrometeorological services was initiated in 1961. In 1961 measurements of total beta-activity of sea water got under way, and in 1962-63 the concentration of Sr-90 started to be measured in the laboratory. We are, therefore, able to follow the dynamics of Sr-90 contamination in western Arctic seas of Russia since 1962. The measurements for Cs-137 were not as frequent as for Sr-90. The analysis for this radionuclide was generally performed during regular surveys on research vessels aimed at investigating radioactive contamination of Arctic sea waters. The first of such surveys was that of the White and Barents seas in 1970 and a comprehensive survey of the Kara sea in 1971. The earlier data for Sr-90 concentration in the Kara sea are shown in Table 1.

Table 1  
Sr-90 in surface waters of the Kara Sea  
in 1963-1964, pCi/l

Coordinates		Month	Year	
°N	°E		1963	1964
76 40	71 02	May	-	0.7
69 48	64 37	August	-	0.5
73 20	65 45	August	1.2	0.5
73 20	80 00	August	1.5	1.3
79 14	74 36	September	0.5	-
75 15	85 21	September	-	2.3

Data of systematic observations of Sr-90 concentration in the surface waters of the Barents sea at the meridian of the Kola bay to latitude 71°30' are shown in Fig.1. The monitoring data about radioactive contamination of water and bottom sediments in the Barents and Kara sea in the 60s and 70s indicate that even at the time of intense radioactive depositions from the atmosphere resulting from nuclear weapons testing, the levels of sea water contamination with Sr-90 were around a pCi/l or less and presented no danger from a radioecological standpoint. Concentrations of Cs-137 in bottom sediments were rather low

and generally an order of magnitude lower than the level of soil contamination on the coasts of Arctic seas.

The expedition survey of the Norwegian and Barents seas performed by FRG Hydrographical institute in 1972 showed that higher water concentration of Cs-137 occurred within a narrow band along the Norwegian coast and in the southern part of the Barents sea as compared to the open Atlantic and the northern part of the Barents sea [1]. This fact was attributed to inflowing water masses contaminated with radioactive wastes of the Windscale (also referred to as Sellafield) nuclear fuel reprocessing plant. As a consequence, Roshydromet initiated a series of field studies to evaluate the effect that transboundary transport of water masses contaminated with the Sellafield wastes had on radioactive contamination of the western Arctic seas, the Arctic ocean and the Baltic sea [2-5].

In 1978 the expedition of SPA "Typhoon" detected Cs-134 characteristic of the Sellafield in the waters of the south-western part of the Barents sea (See Table 2). The 12th cruise of the research ice breaker "Otto Schmidt" was carried out to study radioactive contamination in the Barents, Kara and Greenland seas and the results confirmed that the radiological situation in the western arctic seas was influenced by the Sellafield wastes. Distribution of Cs-137 concentration in the surface waters in 1982 is shown in Fig. 2 and Table 3 summarizes data on other radionuclides. From the changes in the Cs-137/Cs-134 ratio occurring in the course of contaminant transport from Sellafield to the Barents sea, the transit time of the Sellafield wastes to the western boundary of the Barents sea was estimated. The value was found to be about six years. Using a depth profile of Cs-137 and Sr-90 concentration for the Nordcap-Medvezhy line, annual inflow of these radionuclides to the Barents sea was estimated. On this basis we also estimated the part of Sellafield annual discharges of Cs-137 and Sr-90 going to the Barents sea which equals about 20% of Cs-137 discharge and about 30% of Sr-90 discharge. Estimation of the Sellafield contribution indicates that in 1982 60-80% of Cs-137 in the surface waters in the south west of the Barents sea resulted from the Sellafield wastes.

The studies of 1982 showed that the Sellafield wastes went beyond the study area to the Arctic ocean and White sea. This was the reason why SPA "Typhoon" conducted field studies of radioactive contamination of the White sea in 1984 and of the Arctic ocean using the drifting station "North Pole-27" in 1985-87. The results of these works have been published [4, 5] and we will not dwell on them in this presentation. The dispersal of the Sellafield wastes in the Kara sea is graphically shown in Fig. 3. It can be seen that the contaminated waters in the Barents sea go below fresh surface waters of decreased salinity and move north-east at the depth of 50-100 m. In 1982, in addition, samples of bottom sediments were collected and gamma-spectrometric analysis was performed. Table 4 gives density of Cs-137 contamination of bottom sediments. No gamma-emitters other than Cs-137 were detected in the samples.

As is seen from Fig.4 showing change in the Sellafield Cs-137 discharges [6,7], the discharge of the radionuclide is dozens of times lower than the highest value in 1975-1978. Consequently, as the Sellafield wastes were the dominant contributor to the Cs-137 contamination in the south Barents and

Kara seas, levels of activity in these sea regions must have been significantly reduced as compared to those occurring in 1982 (practically to the levels associated with global radioactive background). This was actually supported during the Russian-Norwegian expedition in 1992 in the Kara and Barents sea and the preliminary results of the cruise suggest considerable reduction in Cs-137 levels in the surface waters.

In the course of monitoring of radioactive contamination of the marine environment in Arctic seas, data were also collected permitting inputs of various sources to radioactive contamination of the marine environment to be assessed.

We will focus on Cs-137 and Sr-90 inputs to the Barents and Kara seas from rivers, atmospheric depositions and the Sellafield wastes. The amount of Cs-137 in the soils of the arctic areas in Russia is now about 60 mCi/km<sup>2</sup> and Sr-90 - about 40 mCi/km<sup>2</sup> [8,9]. These values can be used to roughly estimate the amount of the radionuclide in the Barents and Kara seas due to depositions. With the area of the Barents sea being 1438.4 thousand km<sup>2</sup> and that of the Kara sea - 993.4 thousand km<sup>2</sup>, the amount of the radionuclides is estimated at some 86000 Ci of Cs-137 and 58000 Ci of Sr-90 in the Barents sea and 60000 Ci of Cs-137 and 40000 Ci of Sr-90 in the Kara sea. Unfortunately, we have no data about the input of Cs-137 from rivers because the radionuclide was not monitored on a regular basis in outlets of the Russian northern rivers. Yet, data are available from annual observations of Sr-90 in the estuaries of major rivers flowing into the Kara and Barents seas. The amount of Cs-137 can thus be assessed from the Cs-137/Sr-90 ratio in the river water and the value has been found to be 0.1 and less. From the observations of Roshydromet, the amount of Sr-90 transported by the waters of major rivers (Ob, Enisey) to the Kara sea over the period 1961-1989 is about 30000 Ci. Because of a good water exchange between the White and the Barents seas, it can be assumed that Sr-90 which enters the White sea with rivers is, for the most part, carried farther to the Barents sea. Under this assumption the amount of the radionuclide carried with the waters of the major rivers (Pechora, Onega, Severnaya Dvina) over the same period is about 5500 Ci. Taking the Cs-137/Sr-90 ratio in the river water to be 0.1, we get that the amount of Cs-137 in the Kara sea is about 3000 Ci and in the Barents sea - about 550 Ci.

The published data are that since the Sellafield plant was brought into operation, more than 1000000 Ci of Cs-137 [6,7] has been discharged in the Irish sea. Unfortunately, we do not have complete information about the dynamics of Sr-90 discharge in Sellafield. According to [6] the Sr-90/Cs-137 ratio in the discharge is about 0.15 and, hence, the total discharge is about 150000 Ci for Sr-90. As was mentioned above, our estimates of the fraction of the Cs-137 discharge entering the Barents sea is about 20% and about 30% for Sr-90, i.e. the total amount is about 200000 Ci of Cs-137 and about 45000 Ci of Sr-90. We were not able to do similar estimation for the Kara sea because of the lack of data on the water exchange through the Novaya Zemlya straits. The total input of Sr-90 and Cs-137 in the Barents and Kara seas is thus estimated at:

	<u>Cs-137</u>	<u>Sr-90</u>
Deposition	150000 Ci	100000 Ci
River flow	3550 Ci	35500 Ci
Sellafield input	200000 Ci	45000 Ci

Summing up the results of monitoring of radioactive contamination in the Barents and Kara seas, we can refer to the radiological situation in the study sea areas as normal. Over the monitoring period, the levels of radioactive contamination of water in the study areas were much lower than the standards existing in Russia [10,11].



Table 2

Results of 1978 survey of the radioactive contamination of the Barents Sea waters (summer).

Coordinates		Layer	Cs-137	Cs-134	Sr-90	Tritium
°N	°E	M	Bq/m <sup>3</sup>	Bq/m <sup>3</sup>	Bq/m <sup>3</sup>	T.U.
69 30	33 30	0	18.1		8.5	
71 30	33 30	0			10.4	
72 20	33 30	0	14.1	0.55	8.9	
72 20	33 30	100	13.2		8.9	5.2
72 20	33 30	250	8.4		6.3	6.6
72 30	35 00	0	15.9	0.48	7.4	
72 30	35 00	100	17.2		8.1	
72 30	35 00	255	10.2		9.2	
71 50	39 00	0		0.37	6.7	5.0
71 50	39 00	100			7.0	3.4
73 30	44 30	0	9.3		7.4	5.2
73 30	44 30	100	10.6		7.4	3.3
73 30	44 30	320	6.6		6.7	5.2
75 55	46 00	0	8.8		5.2	6.5
75 55	46 00	100	8.4		9.3	7.3
75 55	46 00	300	5.3		8.5	1.1
74 30	47 00	0	9.7		7.0	5.3
74 30	47 00	190	11.0		8.1	5.7
71 30	46 27	0	10.7		6.3	9.2
71 00	51 48	0	20.7		8.1	6.1
68 11	40 28	0	15.0		8.1	8.1

Table 3  
Amount of Cs-137, Cs-134 and Sr-90 in the surface waters of the Barents, Kara and Greenland seas in  
August-October 1982.

Station Number	Sampling Date	Concentration, Bq/m <sup>3</sup>			Ratio	
		Cs-137	Cs-134	Sr-90	Cs-137/Cs-134	Cs-137/Sr-90
7-37	10.08	29	0.6	14	48	2.1
6-16	11.08	27	0.5	10	54	2.7
42	12.08	23	0.4	12	57	1.9
48	14.08	18	0.2	10	90	1.8
47	14.08	27	0.5	11	54	2.5
46	15.08	24	0.6	8	40	3.0
45	15.08	22	0.5	8	44	2.8
51	17.08	3.4	-	7	-	0.5
52	17.08	4.6	-	9	-	0.5
53	17.08	6	-	9	-	0.7
54	19.08	9	0.08	8	110	1.1
55	21.08	7	-	20	-	0.35
56	22.08	17	-	21	-	0.8
6A	24.08	8	0.07	-	110	-
7-35	24.08	13	-	-	-	-
2-26	25.08	15	-	9	-	1.7
1-24	26.08	28	-	9	-	3.1
4-18	27.08	32	-	9	-	3.6
1-12	27.08	26	-	-	-	-
50	27.08	30	0.7	-	43	-
20-37	28.08	26	0.5	7	52	3.7
57	28.08	22	0.5	8	44	2.8
6-6	29.08	33	0.7	10	47	3.3
58	30.08	37	0.9	11	41	3.4
4-3	31.08	24	0.5	6	48	4.0
12-3	01.09	17	0.2	7	85	2.4
6-19	03.09	8	-	3.6	-	2.2
8-19	04.09	6	-	3.4	-	1.8
1-140	04.09	12	-	6	-	2.0
7-140	06.09	9	0.09	3.5	100	2.6
17-140	08.09	9	-	5	-	1.8
22-140	08.09	8	-	3.6	-	2.2
36	12.09	6	-	4.6	-	1.3
60	15.09	5	-	7	-	0.7
61	18.09	4.4	-	4.1	-	1.1
62	02.10	6	-	6	-	1.0
63	03.10	8	-	3.9	-	2.0
64	04.10	4.8	-	4.8	-	1.0
65	08.10	12	-	6	-	2.0
66	09.10	8	-	3.5	-	2.3
67	10.10	11	-	3.0	-	3.7
6-28	10.10	12	-	5	-	2.4
11-28	11.10	13	-	4.2	-	3.1
12-28	12.10	14	-	6	-	2.3
17-6	12.10	18	-	7	-	2.6
15-6	13.10	21	-	7	-	3.0
12-6	13.10	22	-	13	-	1.7
10-6	13.10	25	-	7	-	3.6

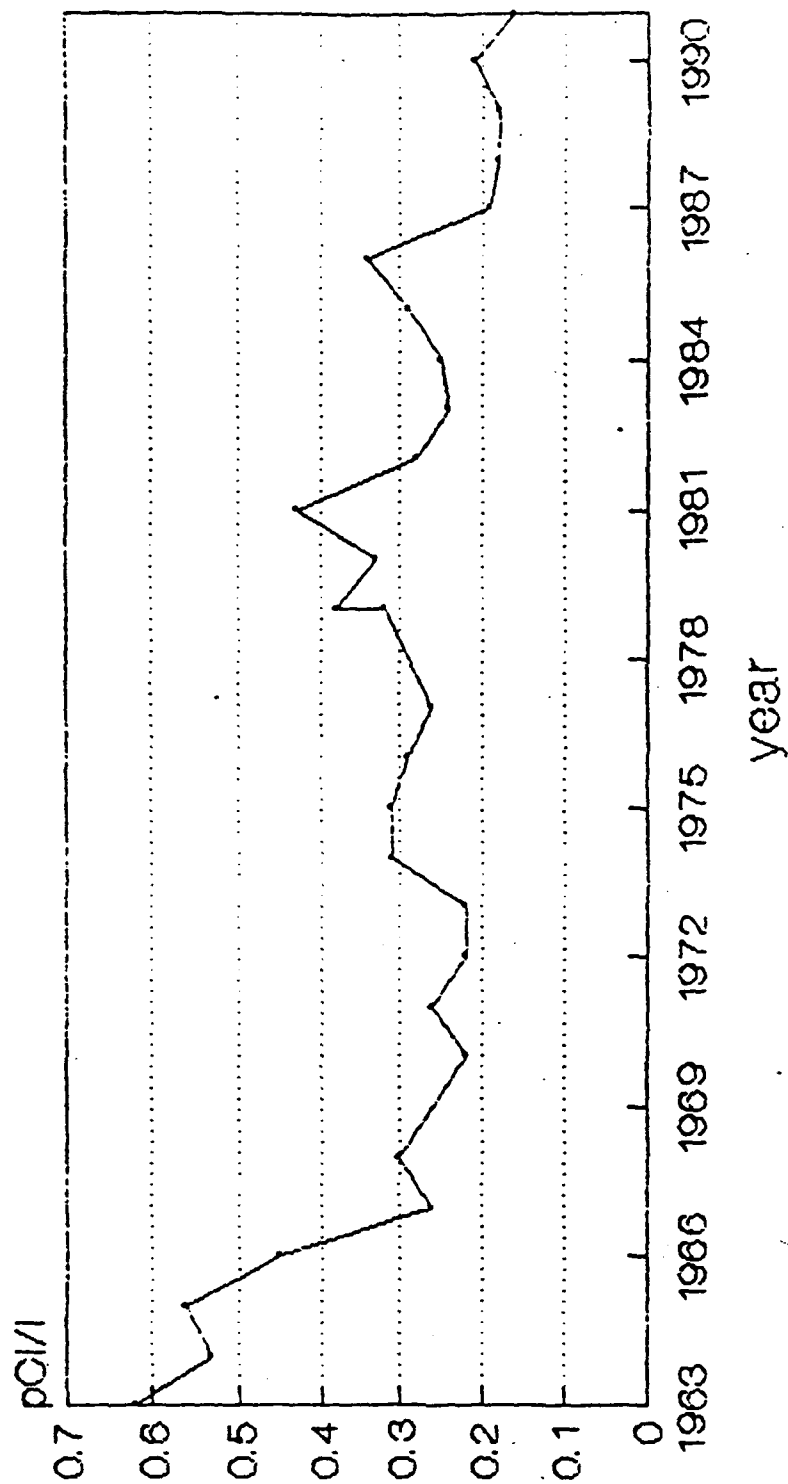
Table 4  
Amount of Cs-137 in the bottom sediments of the Barents, Kara and Greenland Seas in 1982

Station number	Sampling date	Contamination, Bq/m <sup>2</sup>
7-37	10.08	200
6-16	11.08	420
42	12.08	580
48	14.08	100
47	14.08	420
51	17.08	330
52	17.08	390
53	17.08	570
55	21.08	315
56	22.08	305
6a	24.08	210
1-24	26.08	180
4-18	27.08	180
1-12	27.08	280
20-37	28.08	70
57	28.08	65
60	15.09	370
66	09.10	140
12-28	12.10	130

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# Sr-90, Barents Sea Kola meridian, up to 71 30 n.a.



--- Surface water

*Fig. 1*

1982 · 12th trip of research ship "Otto Schmidt"

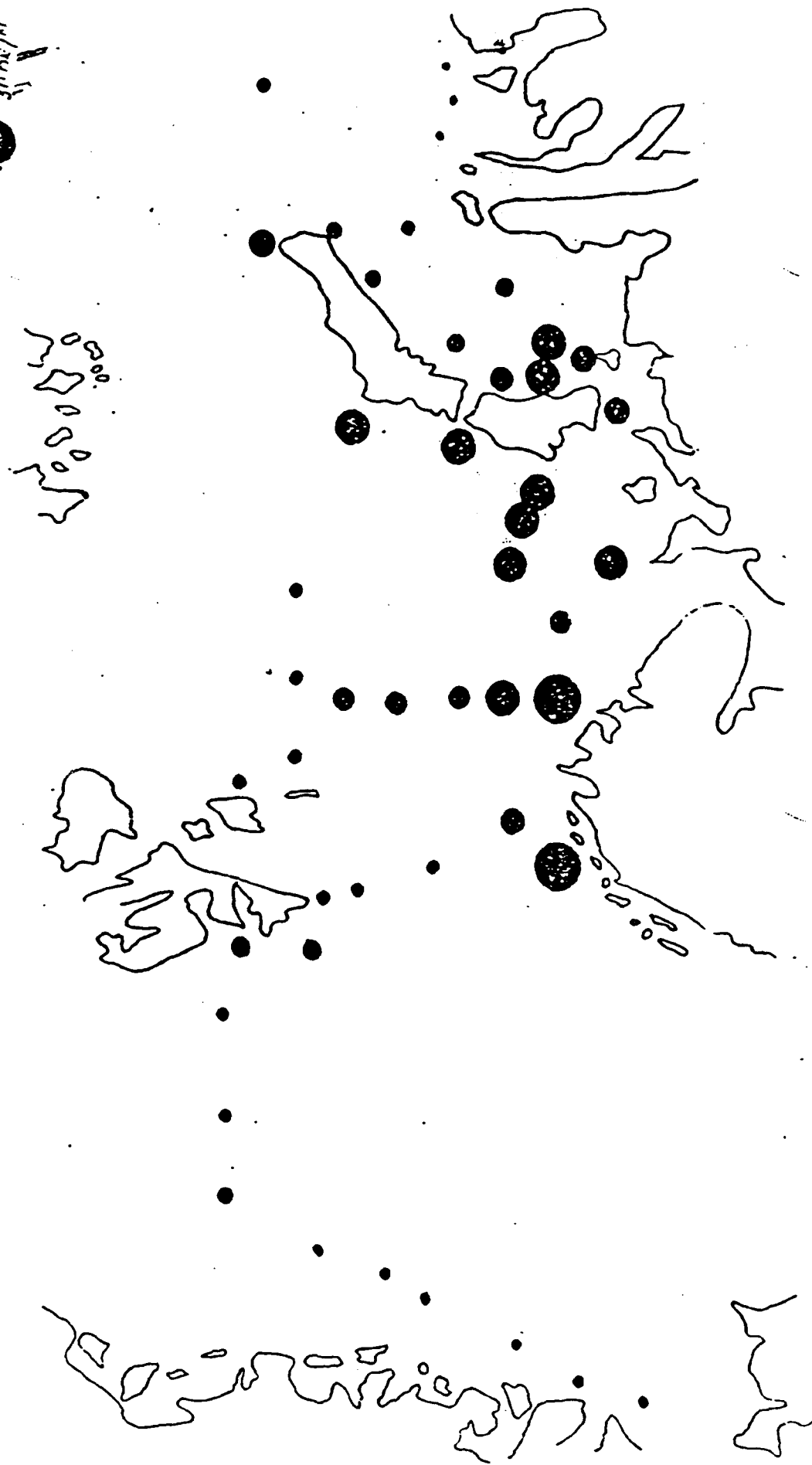


Fig. 2 Location of sampling points in Arctic seas and distribution of  $^{137}\text{Cs}$  concentration in surface waters (circle diameter is proportional to concentration). The points coincident with standard hydrological cuts indicate the number of the point (left) and the number of the out-right.

1982 12th trip of research ship "Otto Schmidt"

Karskiye vorota Str.  
г.р. Карские ворота

Is. Severnaya Zemlya  
о-в Северная Земля

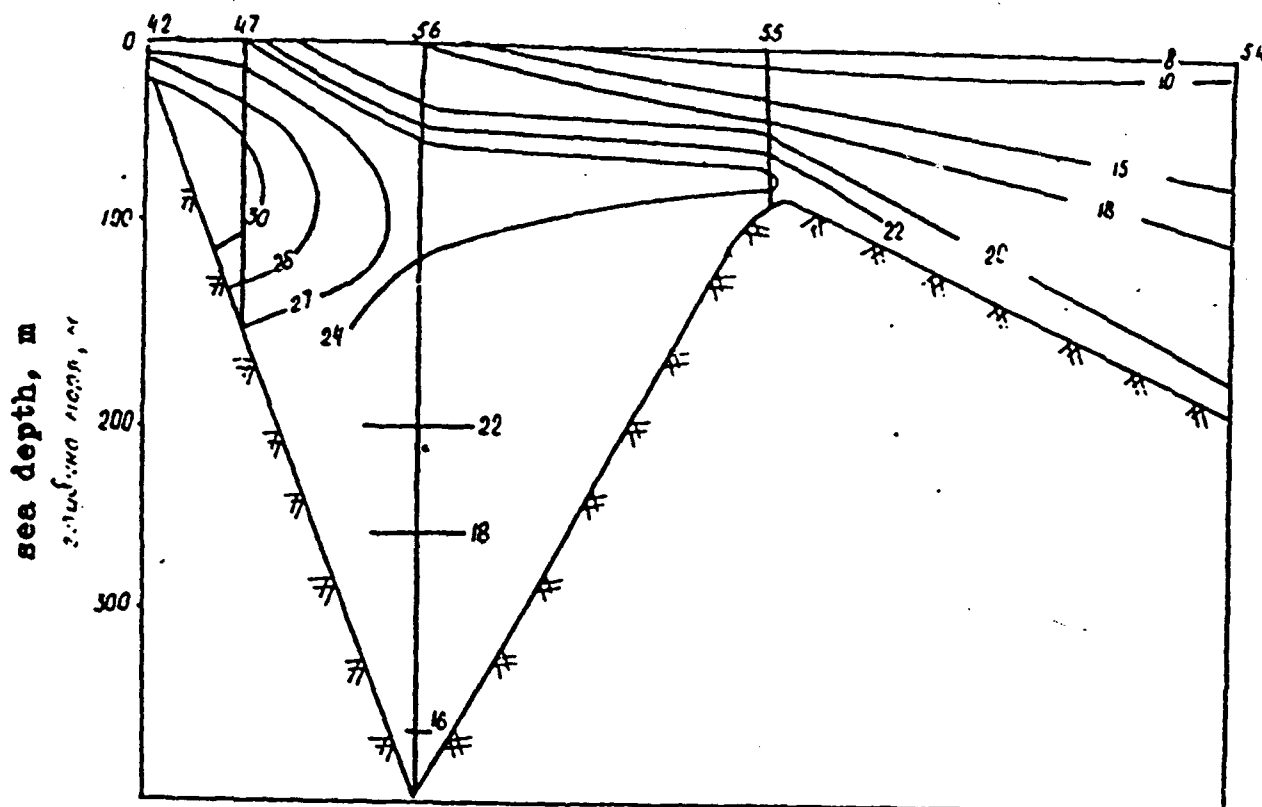


Fig. 3. Isolines of  $^{137}\text{Cs}$  concentration ( $\text{Bq/m}^3$ ) in the Kara sea for the cut Karskiye vorota Str-Severnaya Zemlya

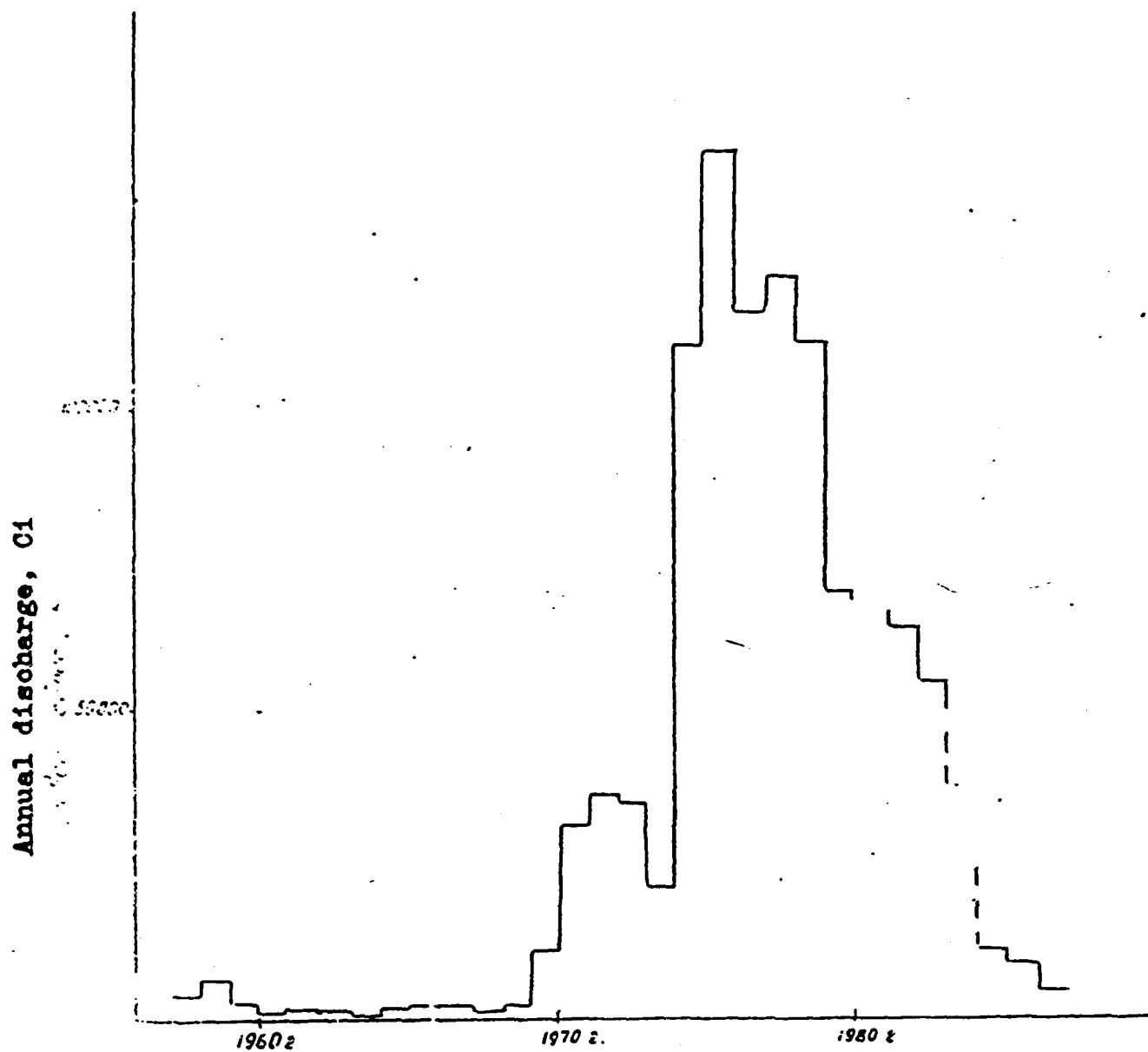


Fig. 4. Dynamics of Cs-137 discharge to the Irish Sea in Sellafield



## **Arctic Radioactive Contamination: Past Legacy and Future**

Presented at the International Conference  
"Radioactivity and Environmental Security in the Oceans:  
New Research and Policy Priorities in the Arctic and North Atlantic", June 7-9, 1993.  
at Woods Hole Oceanographic Institute, Woods Hole, Massachusetts, U.S.A.

Dr. Vladimir Iakimets,  
Institute for Systems Analysis,  
Russian Academy of Sciences

### **Introduction**

This presentation is based on results of the team that prepared the report on dumping radioactive waste by the U.S.S.R. over the last 35 years (1), at the request of President B. Yeltsin. To develop a comprehensive overview of all documents on Soviet solid radioactive waste (SRW) dumping and liquid radioactive waste (LRW) discharge into the oceans, the following cases had to be considered:

- routine operations of the Navy and civilian transport ships for low level LRW discharge into the seas;
- intentional dumping of low and medium level SRW, according to internal regulations;
- intentional dumping of high level SRW (submarine reactors with spent nuclear fuel (SNF));
- accidental SRW dumping due to submarine collisions, wrecks, etc.;
- nuclear warheads lost;
- lost satellites, radioisotopes thermoelectric generators (RTGs), etc.

However much of this is beyond the scope of this report, and many extensive publications are available on accidents at sea (2-4).

Therefore, the attention of this report will focus mainly on LRW discharge and SRW dumping by the Soviet Navy and by vessels of the Ministry of Transportation for the last 35 years.

This presentation has four main objectives:

1. Outlining the findings of the Yablokov Commission on U.S.S.R. RW dumping.
2. Summarizing the past legacy of dumping and describing the options for alleviating possible harm to marine life, the environment and people.
3. To outline the main problems of the Russian Navy for the transitional period.
4. To identify research tasks that will be needed in the years to come.

### **I. Yablokov Commission Findings**

According to a special decree signed by President Yeltsin, a governmental Commission on issues connected with dumping radioactive wastes at sea, was established on October 24th, 1992. Professor A. Yablokov, the President's Ecological Advisor, was appointed the leader of this Commission.

In order to obtain comprehensive information on RW dumping at seas adjacent to Russia's territory, top-ranking representatives of all ministries and departments involved in activities involving RW disposal were introduced into this Commission, including the Ministries of Environmental Protection, Defense, Foreign Affairs, Health Care, Atomic Energy, Transport, and Security, as well as the Russian Committee for Defense Industries, the State Nuclear Inspectorate, the State Committee for Sanitary and Epidemiological Surveys, the State Meteorological Committee, and the Russian Supreme Soviet.

Within the framework of the Commission, a working group consisting of four teams and an expert group, was created (Figure 1).

Also, many specialists from the Archangelsk, Kamchatka and Murmansk regions and from Primorsky Krai, were involved in the Commission's activities.

Four main objectives were developed at the first meeting of the Commission:

1. Collecting and exposing all archival federal documents and materials from executive bodies on RW dumping into the seas by U.S.S.R. and Russia;
2. Studying international legal documents and legal aspects of the problem under consideration;
3. Creating an initial inventory on RW dumping and discharge into the seas by the U.S.S.R.;
4. Analyzing the consequences of RW dumping practices on the marine environment and economy.

Background information (including previously classified data) was delivered to the Commission by several federal executive bodies and the above-mentioned regional authorities. In November 1992, the Main Staff of the Russian Navy submitted valuable official data on RW dumping and disposal practices.

It is necessary to point out that before this time, no single official document had been made available. The first time information on Soviet RW dumping into the seas was leaked to the press was in 1990, due to the activities of several People's Deputies of the USSR. The ecological grass-roots movement "Towards a New Earth" widely distributed that information. It should be stressed that the first public statement was made by the former U.S.S.R. People's Deputy, Andrei Zolotkov, a radiologist from Murmansk Shipping Company. He made this disclosure at the press conference organized by Greenpeace on September 23-24, 1991 (5). This unofficial data was also distributed by environmental activists. A series of articles and press releases by Greenpeace deserves special mention, as these triggered the activities of many specialists outside Russia. In the information submitted by the Russian Navy, many of the previously disclosed sites of RW dumping in the Kara and Barents Sea, were confirmed, as well as a description of operations on dumping submarine reactors with high level radioactive waste (SNF), containers with low and medium level RW, barges and ships.

The information received by the Commission as a result of official trips to the regions, and extractions from archival documents provided the groundwork for making the following findings:

1. An initial inventory was created of LRW and SRW dumped into the Barents and Kara Seas in the Arctic, and into the Sea of Japan and into the Pacific near Kamchatka, (Table 1), including:

- a. low level LRW discharge in five vast officially designated areas of the Barents Sea, west of the Novaya Zemlya Archipelago, in one area of the Kara Sea, and in five places along the coastline of Russia's Pomorie due to accidents, as well as in six sites in the Sea of Japan and in three areas near the East Coast of the Kamchatka Peninsula in the Pacific (Table 2).
- b. low and medium level SRW dumped in containers, inside of sunken ships and in an unpacked form in 8 sites of the East Coast of Novaya Zemlya, in 3 places of the southern part of the Barents Sea, as well as in 4 sites of the Sea of Japan and to the East from Kamchatka (Table 3).
- c. High level SRW in the form of submarine reactors were dumped into five sites from 1965-1988, along the east coast of Novaya Zemlya and in the Novaya Zemlya Depression, including 6 reactors with unloaded SNF, 1 reactor screen with 125 irradiated fuel assemblies, and 10 reactors without SNF (Table 4). Two other reactors of submarines without SNF were dumped into the Sea of Japan.

It is necessary to clarify that the archival documents and official sources on SRW dumping, considered by the Commission, contain data on SRW activity measured by an artificially designed indicator - activity (equivalent of  $^{90}\text{Sr}$ ), curies. This indicator has been described in "Regulations for Radioactive Waste Dumping at Sea". Numerical values of this indicator were determined on the basis of the measurement of the dose intensity near the SRW volume subject to dumping, by using a simplified empirical relationship based on a priori information on radionuclide content in that volume.

Therefore data on activity of SRW given in Tables 1,3 and 4 and further in the report, could be used for illustrative purposes.

Data in Table 1 shows that activity of the low level LRW and the low and medium level SRW dumped by the U.S.S.R. in the Arctic is twice higher than in the Pacific. One can clearly see this from Table 2, which contains data on sea-wise breakdown of LRW discharge. Besides the single LRW discharge with an 8,500 Ci activity into the Kara Sea in 1976 from the icebreaker Lenin, two thirds of the total LRW activity was discharged into the Barents Sea during the period from 1960-1992.

As can be seen from Table 3, the Kara Sea became a major SRW dump site for the Soviet Union. Of the total SRW dumped, about 70% of the activity resulted from being dumped in 8 inlets there officially designated for that purpose, on the east coast of Novaya Zemlya and in 3 other sites for the period from 1964-1991.

The Kara Sea also received the most environmentally dangerous part of the SRW, namely reactors of nuclear powered submarines with unloaded SNF and their activated components, with a total activity of about 2,400,000 Ci. According to statistical information on accidents, 9 nuclear reactors and about 50 nuclear warheads have sunk into the ocean. These were accidentally "dumped" into the oceans. Reactors in the Kara Sea, on the other hand, were dumped intentionally.

Table 4 contains a detailed description of sites where reactors with SNF aboard, as well as reactors without SNF, were dumped. It is important to stress, nevertheless, that all five objects (with six reactors and one reactor screen) containing unloaded SNF, were dumped after states of emergency which made the removal of SNF impossible before dumping. Some of these reactor compartments and their components were filled with furfural mixtures before dumping; others were additionally placed either in reinforced concrete or metal containers, with or without lead shells.

Unfortunately, the actual radionuclide content of the objects dumped were not available to the Commission, except for the reactor screen of the OK-150 unit from the icebreaker Lenin. Therefore total activity for all submarine reactors has been estimated by experts. For comparison, in Table 4 the Commission's estimates are given, along

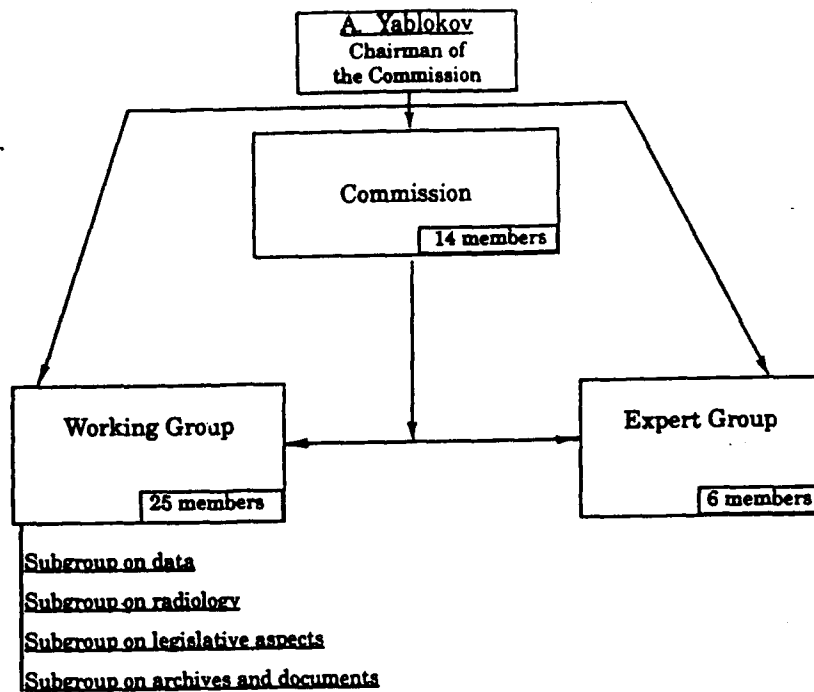


Figure 1 Organizational Structure of Commission

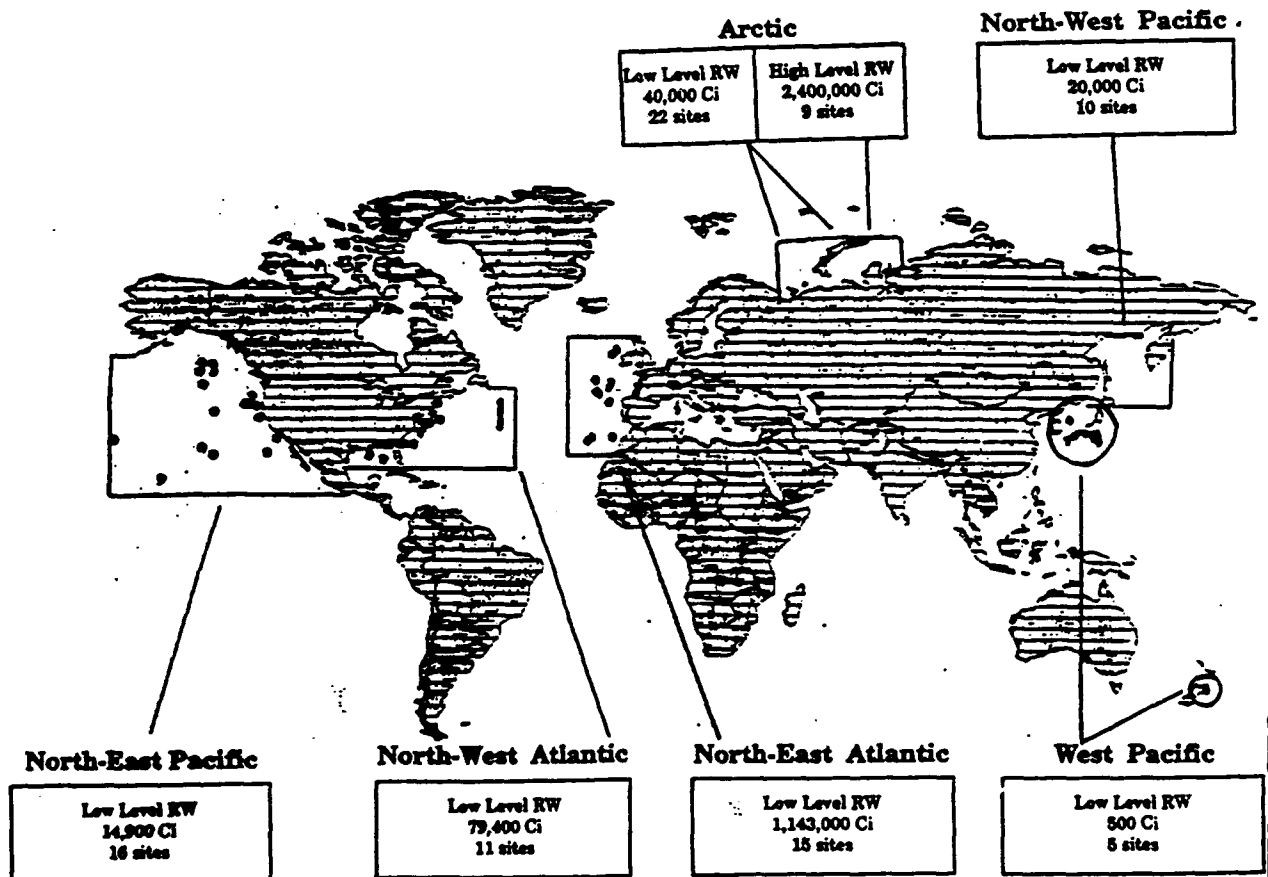


Figure 2. Update of the IAEA data on sites and quantities of radioactive waste disposal in the oceans

with recently published estimates made by the Lawrence Livermore Lab scientists on the basis of a computer model ORIGEN 2. One can see that for all objects but one, the estimates for the activity at the date of disposal, are reasonably close to each other. The main difference is related to estimates of the activity of three reactors and a screen assembly of the icebreaker Lenin OK-150 unit. We think that the LLL scientists overestimated the activity of this object at the time of disposal. The upper estimate of activity of 4,170 kCi is higher than it should be. If both the lower and upper estimates for the OK-150 are subtracted from corresponding total estimates, then a range of total activity for fission products would be within the range of 1,000-3,300 kCi, which would fit with the findings of the Yablokov Commission.

2. Normative documents and acts developed by the U.S.S.R. Navy and several ministries for regulating radioactive waste disposal, were in conflict with international norms and rules. In fact, all sites used by the U.S.S.R. for RW dumping, according to "Regulations on RW Discharge into Seas" were located on territories, where RW disposal had been strictly prohibited by the London Dumping Convention. Figure 3 updates the IAEA data on the sites and quantities of RW dumping, based on the Yablokov Commission Report findings. In fact, two vast new regions where radioactive wastes were dumped over a period of more than 30 years, have been added to and noted on a map compiled by the IAEA, namely:

- Arctic Region, located in the higher latitudes, with 31 sites, and
- North-West Pacific Region, located in the Russian Far East, with 10 sites.

Figure 3 shows that RW dumping into Arctic seas by the Soviet Union was against the 3 main requirements of IAEA. Looking at the North-West Pacific Region, one can see that the only legally acceptable place where Russia could continue RW dumping if a corresponding decision is made, is located in that region.

Comparative data on dumping for all regions is given in Table 5. Comments given in this table show that an IAEA inventory is quite far from completion.

Furthermore, dumping of reactors with SNF into shallow Arctic Seas, can in fact be considered as a criminal violation of international law. As a result, inlets of the Eastern Coast of Novaya Zemlya, were turned into an uncontrolled radioactive dump.

3. The sites in the Kara Sea and in the Sea of Japan (in close vicinities to the dumping sites of the reactors) have not been monitored for radioecological conditions in the last 25 years.

All investigations of radioecological situations have been conducted in areas that were located 50-100 Km from the SRW dump sites. In Figure 4, the routes taken by 3 expeditions on the research ships "Otto Schmidt" (1982), "Victor Buinitsky" (joint Russian-Norwegian expedition 1992), and "Dalnie Zelentsy" (1992), for the purpose of studying radioactive contamination of both the Kara and Barents Seas, are shown on the map by a dotted line, a broken line and a continuous line correspondingly.

The Roman numerals on this map show the sites for SRW dumping. Nine sites where reactors of nuclear-powered submarines and the icebreaker Lenin were dumped are shown by squares with Arabic numerals. An interesting observation was made by Yablokov, by examining the results of the international expedition (1992). He found that for at least five sample points (encircled on the map) taken at different depths,  $^{137}\text{Cs}$  concentration (Bq/cubic meter), along the Novaya Zemlya Depression, increases as you move from the surface to the bottom of this depression (Table 6). These findings show there is a need for a new expedition to provide more data for statistical analysis, in order

Table 1. Yablokov Commission findings on RW dumping

	<u>Arctic</u>	<u>Pacific</u>	<u>Total</u>
1. SRW			
sites	11	4	15
ships	17	38	55
containers	6508 +	6642 +	- 13,150
bulky objects	155	100 +	255 +
Volume, cubic meters	31,534	21,842	- 54,000
Activity, Ci ( $^{90}\text{Sr}$ )	15,900	6,700	- 22,600
2. LRW			
areas	5 + 6	9	14 + 6
Stock, cubic meters	190,000	123,500	313,500
Activity, Ci	12,100 + 11,600	12,400	36,100
3. Reactors with SNF			
sites	5	--	5
number of	7	--	7
Activity, Ci ( $^{90}\text{Sr}$ )	2,300,000	--	2,300,000
4. Reactors without SNF			
sites	5	2	7
number of	10	2 + screen	12 + screen
Activity, Ci ( $^{90}\text{Sr}$ )	100,000	116	100,100

Table 2. L R W Discharge by U.S.S.R.

<u>Area</u>	<u>Activity Ci</u>
1. Barents Sea	15,171
2. Kara Sea	8,500
3. White Sea	100.02
4. Baltic Sea	0.2
	Subtotal: - 23,800
5. NW Pacific (east of Kamchatka)	352.2
6. Sea of Japan	11,984.4
7. Sea of Okhotsk	0.1
	Subtotal: - 12,340
8. Discharge from Ob and Yenisei Rivers	Several thousand

TOTAL: 36,140 + (n x 1,000) ~ 40,000

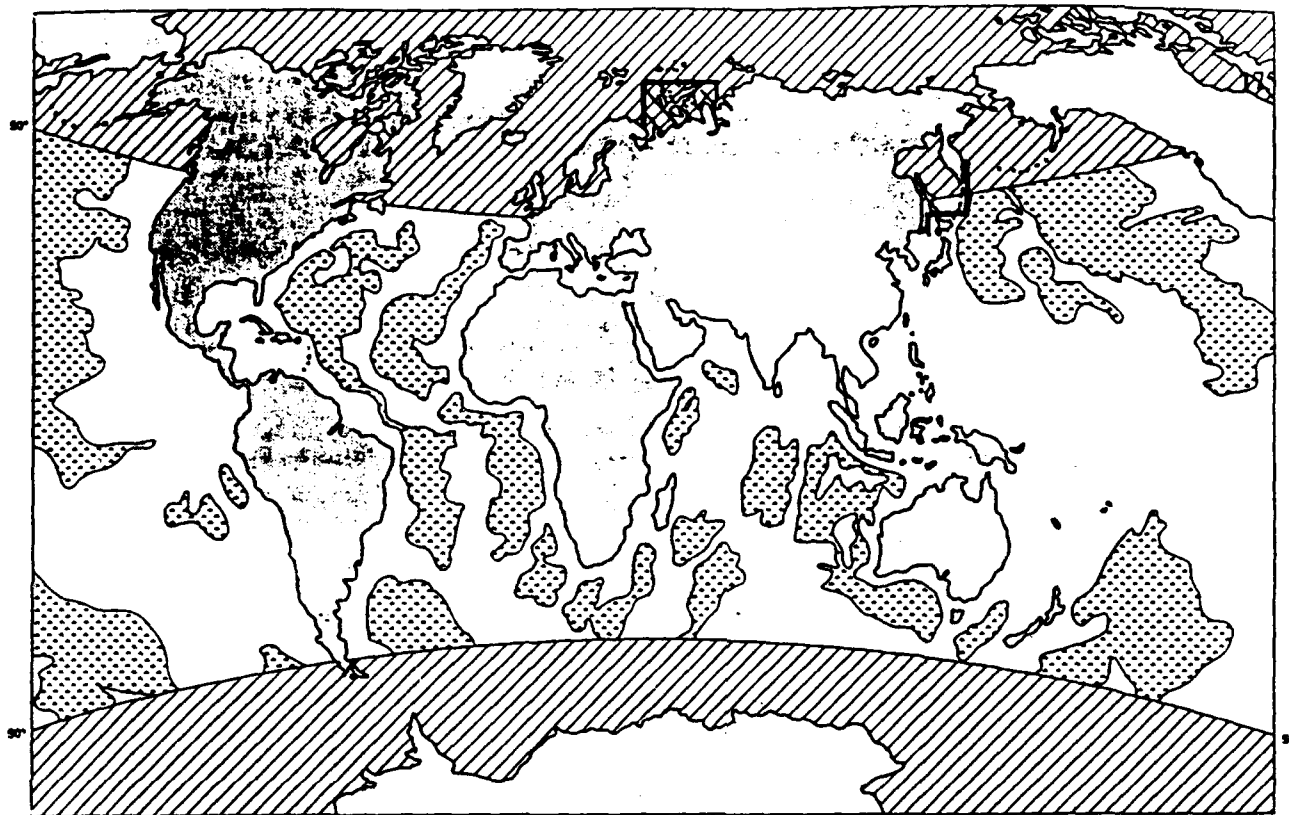


Figure 3. Two Soviet dumping regions shown on the map with zones in oceans open for dumping of low and medium level RW (IAEA)

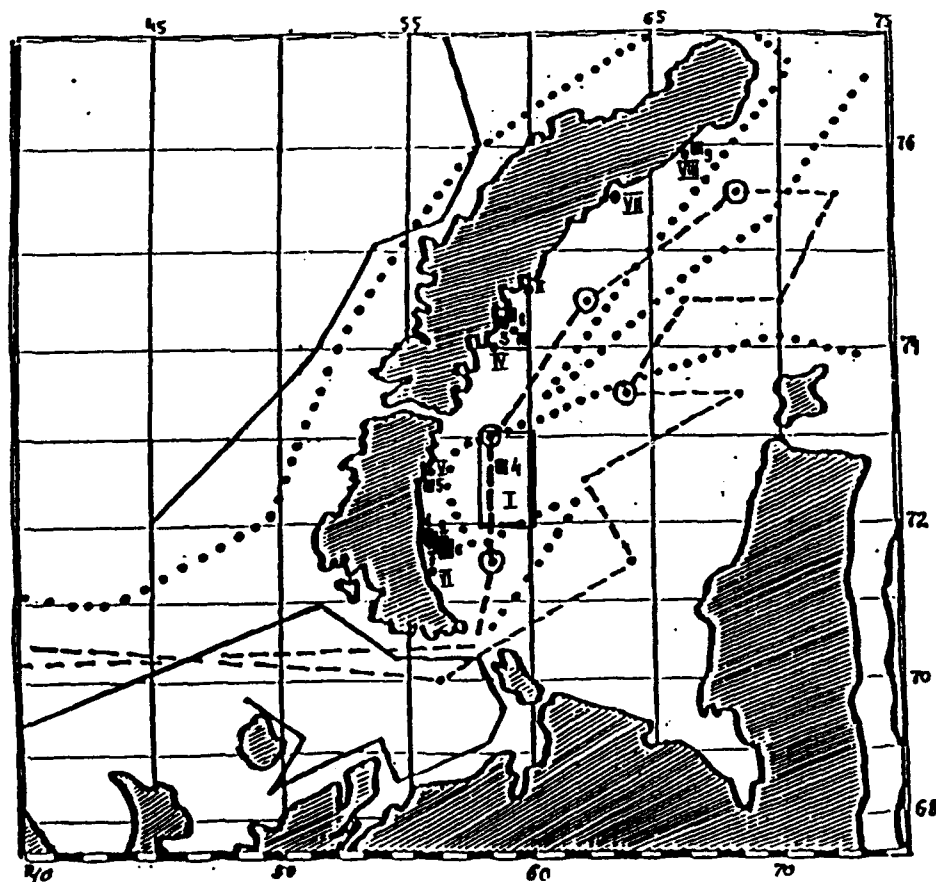


Figure 4. Sites of SRTW dumping and routes of research expeditions for studying radioactive contamination in the Kara Sea

Table 3. S R W Dumping by U.S.S.R.

		Activity Ci ( <sup>90</sup> Sr)	
		Without reactors	Sub reactors
<b>INTENTIONAL DUMPING</b>			
1	Barents Sea	440	
2	Kara Sea		
	-containers, ships and objects	15,462	
	-reactors with SNF (7)		2,300,000
	-reactors without SNF (10)		100,000
	Subtotals in Arctic:	≈ 16,000	≈ 2,400,000
3	NW Pacific (Kamchatka)		
	- containers, ships and objects	2,922	
	-1 shield of sub's reactor		70
4	Sea of Japan		
	-containers, ships and objects	3,774	
	-2 reactors without SNF		46
	Subtotal in Far East:	≈ 6,700	116
	Total for Arctic and Far East:	≈ 22,600	≈ 2,400,100
<b>ACCIDENTAL DUMPING</b>			
5	Sunken nuclear subs		≈ 650,000
6	Lost NWH's, RTG's, Satellites	n x 1,000	
	Totals:	≈ 30,000	≈ 3,050,100

Table 4 Reactors dumped into the Kara Sea and their characteristics

#	Disposal Date	Site, Coordinates	Number of subm. (orders)	Number #	Core	Expert estimates kCi at disposal date	Estimates by LLL, 1993 kCi	
							at disp. date	at present
1.	1965	Abrosimov Inlet 71° 56' 02" N. 55° 18' 05" E.	285	2	1	800 + 10	663 - 2,300	195 - 213
2.	1965	71° 56' 04" N. 55° 18' 15" E.	901	2	2	400		
3.	1965	71° 55' 22" N. 55° 32' 54" E.	254	2	--	- 10		
4.	1966	71° 56' 03" N. 55° 18' 08" E.	260	2	--	- 10 (subtotal = 1230)		
5.	1967	Tsimlka Inlet 74° 22' 02" N. 58° 42' 04" E.	OK - 150 Screen assembly	1	1.8	100	1,990 - 4,170	262 - 269
6.	1967	74° 26' 06" N. 58° 37' 05" E.	(60% of SNF) OK - 150	3	--	50 (subtotal = 150)		
7.	1972	Noraya Zemlya Depression 72° 40' N. 58° 10' E.	421	1	1	800	213 - 811	80 - 86
8.	1981	Stenovog Inlet 72° 31' 25" N. 55° 30' 25" E.	601	2	2	200	187 - 191	136 - 139
9.	1988	Tschentva Inlet 73° 59' N. 66° 18' E.	538	2	--	- 10	--	--
Total:				17	7	- 2,400	3,053 - 7,472	673 - 707



to understand the underlying reasons for this phenomena.

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4. Despite both international and U.S.S.R. laws, prohibiting dumping of RW in the seas, the Navy used "the simplest solution to dispose of RW in the seas", as other ways were not properly studied and the generation of RW by submarine fleets and icebreakers from 1959-now "was not considered of primary importance". By the way, this is also true for civilian nuclear power plants.

5. The Russian nuclear-powered Navy has 394 military and 10 civilian reactors. Around 100 of them have been inactivated, 18 of which were dumped at sea. About 80 inactive reactors, containing highly radioactive SNF, are still aboard submarines tied up at moorings along the Russian North Coast and in the Russian Far East.

6. The Russian Navy annually generates up to 10,000 cubic meters of liquid waste and 2,000 cubic meters of solid waste in the Northern Fleet, and around 5,000 and 1,000 cubic meters are generated respectively by the Pacific Fleet. Taken together, both civilian nuclear-powered ships and naval vessels, generate up to 20,000 cubic meters of LRW and up to 6,000 cubic meters of SRW annually. Because of the lack of storage facilities on land, dumping of RW at sea is continuing and will continue through at least 1997.

7. There are about 30,000 highly radioactive spent-fuel assemblies stored in temporary facilities. As their capacities are nearly completely filled, there is a critical situation with respect to the refueling of current operational submarines, not to mention defueling reactors of retired submarines. This should be considered as an emergency situation, as any accident at these facilities may have a strongly negative impact on the environment.

## II. Problem Areas, Options and the Need for a Comprehensive System Analysis

Based on the Commission findings, we can summarize that 5 main issues have to be analyzed in a systematic way to find preferable solutions:

- LRW discharge: when and how it can be discontinued;
- SRW disposal: how and when a risk resulting from the past legacy can be alleviated and what should be done in the future;
- SNF disposal: how and where;
- Naval nuclear reactor compartment disposal and storage;
- Recycling of nuclear-powered submarines and vessels.

None of these issues have been comprehensively studied up to now. There are some estimates by experts on time horizons and budget allocations needed to find and implement proper solutions for the above-mentioned problems. Experts consider, for instance, that a program for discontinuing LRW discharge at sea by the Navy would take five years, and would cost about 1 billion rubles. Their "guesstimates" related to the problem of Naval SRW disposal are even higher.

At present, a state-wide program on RW and SNF management, utilization and disposal for the period of 1991-95 and up to 2005, is still under consideration by the Russian government. It seems to me that the draft of this program may be finalized in 1993, as parliamentary hearings on the State law on RW management have started

Table 5. Comparative data on dumping

All countries, but Russia (IAEA)	<u>Low level RW</u>	<u>High level RW</u>
	Ci ( $^{\circ}$ Sr)	
North-East Pacific	14,900	--
North-West Pacific	79,400	--
North-East Atlantic	1,143,000	--
West Pacific	500	--
Subtotal:	1,237,800	--

- Sewage waters from nuclear fuel processing plants, lost nuclear warheads, sunken nuclear submarines, and radionuclides entering the ocean as a result of underwater nuclear tests were not accounted for. Just the LRW from Sellafield and other U.K. and French plants are estimated at 1,000,000 Ci.
- Data from China, Pakistan, India, Israel and South Africa are not available.

USSR. (Russia)	<u>Low level RW</u>	<u>High level RW</u>
	Ci ( $^{\circ}$ Sr)	
Arctic Seas (Soviet)	= 40,000	
Reactors with and without SNF		= 2,400,000
North-West Pacific	= 20,000	
Reactors without SNF		116
Total:	= 60,000	= 2,400,000

- Data on LRW discharge into Lake Karachay in the Southern Urals, into the Tom River in the Tomsk-7 territory, and in the Yenisei River near Krasnoyarsk-26 were not accounted for.

Table 6.  $^{137}\text{Cs}$  concentration ( $\text{Bq/m}^3$ ) at various depths at 5 sampling points in the Kara Sea

Depth	Sampling points				
	1	2	3	4	5
Near surface	6.9	2.9	3.2	3.5	3.4
7- 80 meters	5.0	6.9	6.2	10.3	4.4
85 - 320 meters	10.9	11.1	19.5	11.6	9.8

and will continue this summer.

Within the draft of the program, there is a Section 9 "Management of waste, generated in operation and decommissioning of transport nuclear plants". This section contains:

1. A set of proposals for construction of on-shore and ship-borne complexes and facilities for reprocessing LRW and SRW;
2. A plan for conducting in 1993 a statistical survey of RW generation and accumulation in the northern and far eastern regions of Russia, as well as development of technical and economic projects for construction of specialized capacities for temporary RW storage, reprocessing and disposal;
3. A project for construction of pilot industrial storage facilities for SRW and solidified RW of low and medium level activity, in 1993-95;
4. A plan for commissioning in 1996 facilities for storing SRW of high level activity;
5. Proposals for development of projects on decommissioning up to 2000 radioactively contaminated facilities of Naval technical bases;
6. A program for cleaning up and stabilizing the radioactive situation in Chazhma Bay and in Shkotovo-2 (both in the Far East), to be developed in 1993 and implemented up to 1995;
7. A project for estimating the radioecological consequences of LRW discharge, SRW dumping and submarine sinking at sea, including a Russian expedition, with the involvement of foreign specialists, for the purpose of studying dumping sites of submarine reactors with SNF.

The following projects were missing from this program and from the Commission's point of view, need to be included:

1. Immediate removal of SNF from the storage facilities of the Navy and their Murmansk Shipping Company (first of all from floating facilities);
2. Construction of new storage facilities of SNF at the Navy bases;
3. Commissioning of the SNF storage under construction by Murmansk Shipping Company should be shifted from 1995 to 1994.

A set of immediate measures for decommissioning of nuclear-powered submarines and vessels, that are scheduled to be retired by 2000, was included into a decree by the Russian government, adopted in August 1992.

The following were included :

- Construction of sites for a temporary storage of nuclear-powered submarines afloat;
- Refitting of bases/shelters for temporary storage of reactor compartments;
- Construction of on-shore bases for floating service ships, compartments and equipment for reactor cores offloading, reactor defueling, reprocessing of RW, SRW preparation for disposal, sites for temporary storage of liquid and solid RW.

However, the Commission considered that such important issues as the disposal of reactor compartments with unloaded SNF, selection of optimum methods and techniques for storage of compartments, technology for recycling nuclear-powered vessels and warhead compartments were not taken into account by this decree.

In my view, there are three main problem areas that should be clearly separated in order to be able to conduct a systematic analysis of possible options, aimed at choosing

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the most preferable solutions. These problem areas are:

1. Past Legacy;
2. Transitional Period;
3. Long-term Watchdog (this will not be covered in this presentation).

### III. The Past Legacy

This problem area is related to the practices of SRW dumping, LRW discharge, SNF reprocessing, as well as submarine and vessel disposal, that were applied by the Soviet Navy and the Ministry of Transport in the past. Some damage to the environment that occurred at that time is irreversible. The other wounds inflicted on nature can be healed.

A list of tasks that should be considered within this problem area includes, among others, such urgent topics as:

1. Monitoring and lifting off some of the reactors dumped into the Kara Sea;
2. Lifting off the Komsomolets submarine, which sank in the Norwegian Sea (or some of its parts);
3. Decontamination of the Chazhma Bay and the Shkotovo-2 area;
4. Decommissioning and deactivation of old and radioactively-contaminated technical bases of the Navy.

In Box 1, one can find a description of possible options for reactors in the Kara Sea and for the Komsomolets submarine.

Let's consider in detail option #4 for reactors. In May 1993, the computer-estimated inventory of radionuclides in the reactors of submarines dumped in the Kara Sea, was calculated by scientists from Lawrence Livermore Laboratory (6), using a PC program ORIGEN 2. In Table 4, information was presented on the total activity estimates for the dumped reactors.

This inventory can be used as an example of how one of many possible options can be developed for dealing with naval reactors dumped in the Kara Sea. I am not going into details of the computer model, but I would like to use the results of the computations to outline the options. Table 7 contains data on activities in kilocuries (kCi) of selected actinides, fission products of reactors' cores and activation products in the reactor's components. Estimates for primary system corrosion products also given in (6) were omitted from our consideration.

One can see that at the time of disposal,  $^{241}\text{Pu}$  accounted for 97.2% of the actinides' activity,  $^{90}\text{Sr}$ ,  $^{137}\text{Cs}$  and  $^{147}\text{Pm}$  composed up to 19% of the fission products activity, and 100% of activities for the activation products in the reactors' components were related to  $^{60}\text{Co}$ ,  $^{63}\text{Ni}$  and  $^{55}\text{Fe}$ .

Corresponding ORIGEN 2 estimates for 1993 are equal to 84%, 50% and 100%.

Predictions of activities for 2015, given in the right column in the Table (8) show that the main danger for marine life and environment will come from fission products.

While analyzing the results of the ORIGEN 2 computations, I found that the highest percent of total activity of actinides and fission products is generated in the reactors dumped in the Tsivolka Inlet. Namely, 62-99% of the total  $^{241}\text{Pu}$  activity (which gave 97.2% of all actinides' activity at disposal time, and 84% at present) is associated with the Tsivolka Inlet.

Table 7. Inventory of radionuclides estimated with ORIGEN 2

	Date of disposal (1965 - 1988)	kCi	
		1993	2015
• Actinides plus daughters	69 - 109	23 - 38	11 - 18
• Fission products in reactor cores	3050 - 7472	674 - 708	421 - 443
• Activation products in reactor components	916 - 1127	124 - 126	63 - 64
• Activation products in the primary system corrosion products	1.4 - 1.6	0.16 - 0.17	0.0115 - 0.0120

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Table 8. Activity estimates (kCi) for reactors dumped in the Kara Sea (ORIGEN 2, LLL, May 1993)

Radionuclides	Time of Disposal	1993	2015
<b>Actinides :</b>	69 - 109	23 - 38	11 - 18
- <sup>241</sup> Pu (14.4)	67.1 - 106.1 (97.2%)	19.3 - 32.1 (84%)	
<b>Fission Products :</b>	3050-7472	674 - 708	421 - 443
- <sup>90</sup> Sr (28.5)	276.1 - 294.7	157.8 - 168.2	
- <sup>137</sup> Cs (30.0)	306.9 - 318.4	178.3 - 184.7	
- <sup>147</sup> Pm (2.62)	327 - 645 (17 - 30%)	0.6 - 1.2 (60 - 81%)	
<b>Activation Products in reactor components :</b>	916 - 1127	124 - 126	63 - 64
- <sup>60</sup> Co (5.97)	160.6 - 183.6	18.2 - 19	
- <sup>63</sup> Ni (100.1)	81.9 - 82.4	70.1 - 70.6	
- <sup>55</sup> Fe (2.75)	672.8 - 860 (100%)	35.2 - 35.5 (100%)	

Table 9. Contribution of reactors dumped in Tselvolka Inlet to activity (kCi) of reactors dumped in the Kara Sea (ORIGEN 2, LLL, May 1993)

Radionuclides	Total	Inlet of Tselvolka	Percent of total
<b>Actinides</b>			
<sup>241</sup> Pu (14.4) 1967	67 - 106	66 - 69	66-69
1993	19	19 - 32	62-99
<b>Fission Products</b>			
<sup>90</sup> Sr (28.5) 1967	276 - 294	112 - 115	39 - 40
1993	158 - 168	60 - 62	37 - 38
<sup>137</sup> Cs (30.0) 1967	307 - 318	128 - 131	41 - 42
1993	178 - 185	70 - 71	38 - 40
<sup>147</sup> Pm (2.62) 1967	327 - 645	157 - 297	46 - 48
1993	0.6 - 1.2	0.2 - 0.3	--
<b>Activation Products</b>			
<sup>60</sup> Co (5.27) 1967	161 - 184	34 - 38	20 - 21
1993	18 - 19	1.1 - 1.2	6 - 7
<sup>63</sup> Ni (14.4) 1967	82	15.5 - 15.6	19
1993	70	13	19
<sup>55</sup> Fe (2.73) 1967	673 - 860	142 - 183	21
1993	35 - 36	0.20 - 0.25	--

At the time of disposal, around 40-45% of the total activity of fission products ( $^{90}\text{Sr}$ ,  $^{137}\text{Cs}$ ,  $^{147}\text{Pm}$ ) generated by all reactors is estimated to those in this inlet. At present, 50% of all activity is related to  $^{90}\text{Sr}$  and  $^{137}\text{Cs}$ .

For reactors in this inlet, around 20% of the activity of activation products in the reactor's components were due to  $^{60}\text{Co}$ ,  $^{63}\text{Ni}$  and  $^{55}\text{Fe}$  at the time of disposal. At present, their percentage of the total activity of this category is about 15%.

According to the report of the Yablokov Commission, the screen assembly of a reactor from OK-150 unit of the nuclear icebreaker Lenin, with residual SNF (60% of fuel complement based on  $\text{UO}_2$ ), as well as an OK-150 nuclear power plant from this icebreaker, comprised of 3 reactors with primary loop power plants and water-tight stock equipment, were dumped not far from each other, at the depth of 50 meters, in Tsvolka Inlet.

Suppose the assumptions made by the Lawrence Livermore Laboratory scientists, which resulted from the use of the ORIGEN 2 code, as well as their estimates, are reasonably accurate, one can develop the following option for the drastic reduction of possible risk, for the marine environment in the Kara Sea from dumped reactors.

If, for instance, damage to reinforced concrete containers and metal shells of assembly, as well as in biological shielding units of the 3-reactor block are revealed, then a possible option could be a decision to lift the reactors of the Icebreaker Lenin off the bottom of the Tsvolka Inlet. Only this option, when implemented, can help to decrease by roughly half the total activity of radionuclides in the Kara Sea, generated by submarine reactors dumped there, if estimates on activity were made correctly. Completely different options can be suggested for getting rid of the legacy of the past in Abrosimov and in Stepovoy Inlets.

**It is important to stress nevertheless that, for generating a list of possible options within the problem area Past Legacy, historical data on submarines' reactors cores must be declassified and provided to researchers.**

In order to decide on which options are the most preferable ones, a corresponding set of criteria for each of the objects has to be developed. Such criteria as improving safety, expenditures, environmental impact, technological feasibility and compliance with legislation, may be considered as important priorities.

#### **IV. Transitional Period**

If START II, signed in January 3, 1993 by U.S. President Bush and Russia's President Yeltsin, is ratified by the Supreme Soviet, then the Russian Navy will enter the internationally approved process of decreasing its strategic forces down to a reasonable deterrence level. The Navy will also have an opportunity to avoid the mistakes of the past by improving the whole technological system, for servicing its nuclear-powered submarines and vessels at all stages of their lives.

Special attention should be paid to the main stages of the submarine disposal process. This means that as a final objective for the transition period, which will last until 2003, a civilized, environmentally sound and economically efficient system of submarine disposal, starting from activation to recycling, has to be considered.

According to START II, the U.S.A. will limit warheads on sea-launched strategic missiles (SLBMs) to 1700-1750, one third the present U.S. level. Both Trident I and Trident II SLBMs will be equipped with half the number of warheads, as proposed in START I.

Due to the necessity of the restructuring of its nuclear forces, with increasing shift

in favor to SLBMs, Russia may increase its share of SLBMs on submarines from 30 to more than 50 percent. This will give Russia full use of the subceiling of 1700-1750 warheads, however, this doesn't mean that the Russian Navy will have to build additional submarines. On the contrary, more than 30 obsolete submarines, out of the existing 59 will be eliminated.

However, some non-strategic submarines will be replaced by new ones up to 2003. According to Admiral F. Gromov, Commander in Chief of the Russian Federation Navy, a new 10 year ship building program for the Russian Federation Navy has been created, in order to insure that the Navy possesses the optimum deterrent (7).

These messages about future developments of the Russian Navy are important for the purpose of this conference, as they stress that the role of nuclear-powered submarines will increase within the triad of the Russian deterrent forces. Such a restructuring of the Russian nuclear-powered fleet for the transitional period could also be used to improve the management of radioactive waste, generated by the Navy, and improve the system of nuclear submarine disposal.

Let us consider possible options for this transitional period that would decrease the impact of the risk generated by nuclear-powered fleet operations to the environment, especially the risk resulting from applied practices of RW management and submarine disposal.

To do this properly, a number of important features of submarine reactors should be taken into account. Apart from necessary military standards, such as the ability to withstand rapid changes in power level, the ability to withstand combat shock, etc., the overriding importance, to insure that submarine personnel receive no occupational radiation exposure, is that all fission products generated in the fuel during reactor operation, remain within the fuel. The second significant feature of naval reactors we must consider, is related to the lifetime of reactors' fuel. To achieve a long-life reactor core design, for a reasonably compact form, the use of highly enriched uranium was inevitable. Thus, besides containment of all fission products, a considerable amount of highly enriched uranium remains in the reactor core, up to the time of submarine inactivation.

The two above-mentioned features are very significant for our conference, as levels of radioactive contamination in the oceans, generated by Russian nuclear-powered vessels, will in the future depend very much on the safety of nuclear-powered submarines and on their method of disposal.

There are 3 main stages of approved process for disposal of U.S. nuclear-powered submarines (8) (GAO, 1992):

1. inactivation;
2. removal of defueled reactor compartment and its disposal;
3. submarine recycling.

The first stage includes reactor defueling, shutting down all ship systems, removal of equipment identified for reuse, dismantlement of missile compartment (with rejoining the hull if its recycling is supposed to take place later) and preparations for a waterborne storage.

If implemented improperly, a defueling process may cause an accident resulting in radioactive contamination, as happened during the refueling process of a submarine reactor in Chazhma Bay (Russia) in 1985. In other words, the more safe this process is, the less risk for immediate contamination of the environment with radionuclides.

Removal of a defueled reactor compartment includes its disconnecting with all submarine's systems, cutting the compartment from the hull and removing it from the submarine, preparing it for the shipment to the disposal site. Remaining hull sections

have to be rejoined and prepared for waterborne storage or delivered for recycling without this intermediate storage. Disposal of a defueled reactor compartment includes its shipment to storage facilities and placement in controlled storage.

Submarine recycling involves its complete dismantlement and removal of equipment and scrap metals. In figure 5 one can see transformations of a submarine during its disposal. A Status of US Nuclear-Powered Submarine Inactivation Program is shown in Table 9.

Practically the same stages have to be implemented for retired nuclear-powered submarines of Russian Navy. However, due to technological unpreparedness and lack of financial resources, this scheme cannot be fully implemented. In fact, SNF has been offloaded from only 15 percent of the nuclear submarines decommissioned by the Navy up to January 1, 1993. Because the remaining 85 percent of retired submarines are kept afloat, this situation should be considered as of great potential danger for the environment (Figure 6). It is hard to imagine that the complete technological cycle of submarine disposal will not begin until 1997 (which is also a questionable date). Therefore the risk of sinking of at least one of these subs is quite high, as each submarine has many holes in its hull. Furthermore, it is known that if a nuclear submarine with a titanium hull is tied up to an iron mooring for 2-3 months, then the mooring will be rusted completely. A decision was made to use some enterprises of the military-industrial complex in Severodvinsk to recycle the hulls.

The existing capacities of ship reactor defueling systems may provide unloading SNF from the reactors of all retired nuclear submarines only to 2000. The Ukrainian shipbuilders from Nikolaev city stopped building special floating service ships for off-loading reactors with SNF. One of the options for the transitional period could be a solution, developed at the Pacific Fleet and described by Navy Capt. P. Smirnov (9). At the scrap yard at Bolshoi Kamen, after dismantlement of submarine three compartments, including a reactor's compartment and 2 adjoining compartments are cut off. This unit is hermetically sealed and towed to the Pavlovsk nuclear submarine base for dock-side storage (Figure 7). The floating storage of sealed reactor compartments according Capt. P. Smirnov would be less risky to the environment than if a whole submarine remains afloat. He considers that this approach will also save the Russian Navy several billion rubles in reduced maintenance costs. Also, a deep tunnel in the mountains is being constructed as a storage site for reactor compartments of those submarines that will be decommissioned from the Northern Fleet, in the Murmansk region. Specialists believe that that date of its final commission can be shifted from the year 2000 (as was planned) to 1997.

According to data received by Commission up to this time, the Russian Navy has managed to prepare for the long-term storage only 6 reactor compartments.

Defueling of submarines cannot be properly implemented now because of lack of reloading sites and capacities for SNF storage.

As of January 1, 1993 there are about 21,000 spent fuel assemblies stored in the Northern Fleet and 8,400 assemblies in Pacific Fleet. Around 4,500 spent fuel assemblies are stored by the Murmansk Shipping Company on board the floating technical service ships "Lepse", "Imandra" and "Lotta". Their capacities have been fully exhausted. The repository aboard the "Lepse", with activity of 750 kCi is in an emergency condition.

The problem of SNF shipment to reprocessing facilities of the Atomic Energy Ministry has not been resolved due to lack of transshipment sites on both Fleets. Therefore 50 specially constructed containers for shipment of SNF will be used, according to the temporary plans until the transshipment sites are constructed in 1998.

Within the area of Past Legacy, we were dealing with solutions to be found that



## Box 1. Past Legacy

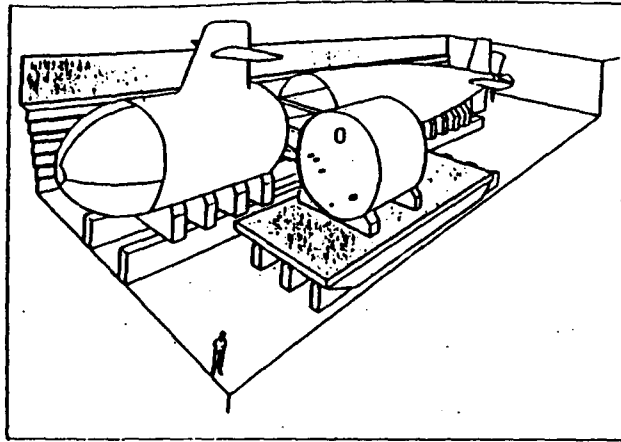
Options	Reactors in Kara Sea	Komsomolets Submarine
1. do nothing	allow them to generate radionuclides and contaminate the sea	allow it to corrode and contaminate the sea
2. watching and registering	monitor, develop models, estimate	monitor, make estimates
3. apply technology available to avert threats	check container status and lift if necessary	fill vessel with gel-forming carbohydrate solution, that would harden around the missiles
4. eliminate the most dangerous part	lift the OK-150 off or clean up the Abrosimov Inlet	lift the torpedo tubes off to prevent the spread of Pu
5. comprehensive solution	lift off most radioactive containers with reactors	lift torpedo tubes off and bury the sub by filling it protective chemical gel or lift the whole vessel

## Box 2. Transitional Period

Options	L R W	S R W	reactor compartments	S N F	subs
1. Do almost nothing.	Discharge at allowed areas.	Dumping at allowed sites.	Keep aboard on floating retired sub.	Keep aboard a floating service ship.	Keep afloat.
2. Temporary solutions to prevent immediate harm.	Collect at sites of generation in containers.	Collect at sites of generation in containers.	Cut off the hull after defueling, seal the unit and store at waterborne site.	Avoid storage aboard a floating ship or in containers open to air, accumulate in special transport containers.	Use some hulls for SRW storage and SNF in specialized containers. Keep others afloat.
3. Intermediate solutions.	Construct facilities for storage of half the annually produced LRW in solidified form.	Construct facilities for immediate transport for storage.	Transfer more than half the defueled compartments to Hanford-type of storage site, or reuse in underground nuclear power station.	Take shipments to reprocessing plants from service ships first.	Recycle at Puget Sound-type dry dock no less than half of the hulls of inactivated subs.
4. Technologically advanced, environmentally sound solutions.	Dispose at controlled land-based storage sites in solidified form.	Dispose at controlled land-based storage sites in containers in compacted form.	Dispose of all at land-based controlled storage sites.	Remove from inactivated subs and reprocess immediately.	Recycle at specialized facilities.

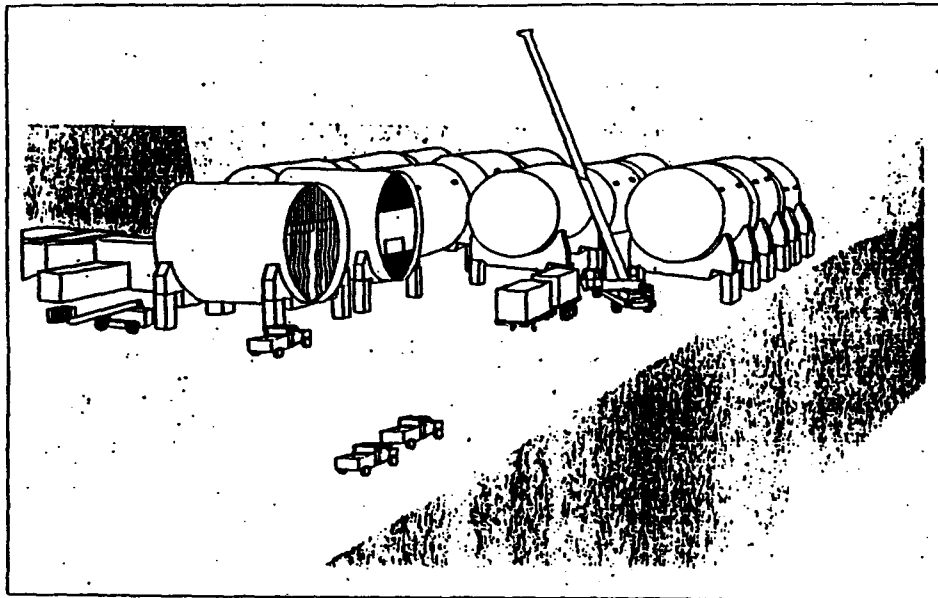
**Figure 5. Removal of defueled reactor compartment and storage of compartments at Hanford (GAO, 1992)**

**Figure 2.1: Inactivated Submarine in Dry Dock With Defueled Reactor Compartment Removed**



Source: Puget Sound Naval Shipyard

**Figure 2.2: Defueled Reactor Compartment Disposal Site at the Hanford Nuclear Reservation, 1990**



Source: Department of Energy, Richland Operations

**Table 10: Status of Nuclear-Powered Submarine Inactivation Program (GAO, 1992)**

Fiscal year	Number of inactivations started	Number of reactor compartments shipped to Hanford <sup>a</sup>	Number of submarine recyclings completed
1969-1980	4	N/A	N/A
1981-1985	7	N/A	N/A
1986	5	1	N/A
1987	4	1	N/A
1988	3	2	N/A
1989	5	2	N/A
1990	5	5	N/A
1991	9	9	2
Total	42	20	2

<sup>a</sup>The first reactor compartment was shipped to Hanford in fiscal year 1986.

<sup>b</sup>The first submarine recyclings were completed at Puget Sound in fiscal year 1991.

would alleviate the impact of accidents with dumped nuclear-powered submarines, RW 187 dumping, mismanagement of SNF, or reactors. In other words, we considered events that happened in the past, which have already damaged the environment or are due to cause damage, if no decisions are made.

The main difference of the Transitional Period area is that we can deal with tasks that will be, or can be implemented, while the Russian Navy shifts from the old practices regarding operation of nuclear-powered submarines to a new more environmentally sound, economically justified and technologically advanced regime.

In fact one can say, that for each significant operation related to servicing nuclear-powered submarines and vessels, the following shifts can be envisaged:

	From	To
SRW	dumping at sea	disposal at controlled land-based storage sites
LRW	discharge at sea	disposal at controlled land-based storage sites with or without solidification
SNF	storage in reactors of subs afloat	immediate reprocessing after removal from sub reactor
S & V	long term ("temporary") waterborne storage	recycling after inactivation, removal of reactor compartments and dismantlement
Reactor compts.	dumping in the Kara Sea or keeping afloat	disposal at controlled land-based storage

In Box 2, possible options for implementing these shifts, within a transitional period, are described.

Let me comment on just one of these proposed options, namely the reuse of nuclear reactors from retired submarines for constructing small underground nuclear power stations. The draft of the engineering project of such stations was approved by the Primorie administration at the end of 1991. It is supposed that the reactors will be placed in the "retired" missile silos. The main reason given for such reuse is the high level of radiation safety. Specialists have said that reactors KN-3 have 5 shielding barriers. However, again, as it was in the past, there are many uncertainties about RW disposal at these stations.

Information on these options recently appeared in the Russian press. An article published in Izvestia on 23 March 1993, showed that the Russian Ministry of Atomic Energy began the development of the second storage site for RW in the Far East, near the town Bolshoi Kamen, in the district Shkotovo. It was mentioned that 27 decommissioned nuclear reactors from submarines in the Pacific Fleet will be disposed there.

However, at the beginning of May 1993, the newspaper Segodnya (Today) revealed that at the discussion held in the Main Navy Technical Department, a new project of refitting 27 (!!) nuclear reactors from decommissioned submarines was being considered for reuse at power stations. The construction of a nuclear power station, in the town Rudnya Pristan, in Terneyskii District, as well as in the Pavloski Bay and on the the Island Gamov, was justified by Navy specialists. This project will cost about 2 billion rubles.



Fig. 6. The conning tower of a scrapped submarine, surrounded by naval refuse at the entrance to Murmansk Harbour.  
September 1990. (c) Greenpeace/Morgan.

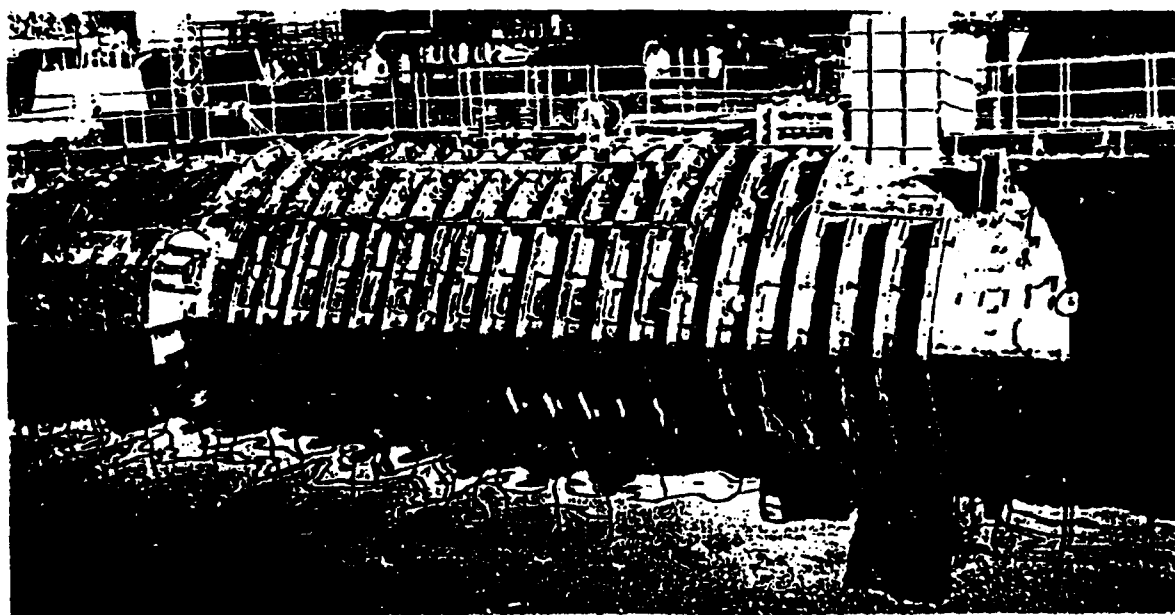


Fig. 7. Reactor vessel of decommissioned nuclear sub stored afloat at the Pavlovsk Submarine Base, near Vladivostok.  
1992, Russian Pacific Fleet, Courtesy of Greenpeace.

## **Conclusion**

The White Book, prepared on the basis of the Yablokov Commission's report to President Yeltsin, can be considered as the first and a very important step in recognizing the true extent of radioactive contamination of the marine environment in the Arctic and the Far East, by the nuclear-powered submarines and vessels.

Due to this Commission's activities, the following actions were taken:

1. Data on all registered sites of LRW and SRW dumped by the U.S.S.R. for the last 35 years in the Arctic and North-West Pacific, was revealed, described and officially made available to the world community.
2. The summed data for the annual or site-wise discharge / dumping of LRW / SRW has been made available for comparative analysis.
3. Cases of violations of international norms on dumping were recorded.

The Commission's findings focused the interest of scientists, diplomats and politicians on the problem of radioactive contamination at sea and possible solutions for improving the situation. Many articles published the conclusions of the White Book. One of the first reactions from the international scientific community was a report of the LLL scientists on reconstruction of radionuclides content in reactors' cores dumped into the Kara Sea. All this is a good sign, indicating that the Commission's results are needed.

It is necessary to mention, nevertheless that uncertainties still exist in the following areas:

- volumes of radioactive wastes dumped;
- radionuclide content;
- geographical coordinates of some early dumping operations;
- activity and composition of SRW dumped;
- activities of reactors with SNF;
- shielding properties of containers and their current status.

I hope that many organizations will join their efforts to continue comprehensive study of the problem. Among the most important topics for consideration, I would list the following:

- System for accounting of LRW and SRW.
- State register for all large objects dumped, including:
  - geographical coordinates;
  - signs for marking dumping sites;
  - maps and navigational aids.
- Comprehensive data base on dumping operations.
- Development of dynamic methods for estimating inventory of radionuclides generated by reactor cores, components, primary systems, and forecasting models of the dynamics of their content.
- Declassification of the normative documents and regulations used for RW disposal at Sea by the U.S.S.R.
- Open access to ships' log-books, registers at naval bases, etc., dealing with RW disposal.
- Monitoring the radioecological situation in areas of RW dumping.
- Development of projects for lifting off dangerous objects.

- 190 o      System analysis of options for finding preferable solutions for the transitional period.

Finally, on April 26th 1993, President B. Yeltsin signed an act requesting the Ministries of Environmental Protection, Foreign Affairs, Atomic Energy and Roshydromet, in cooperation with the Interagencies Commission on Arctic and Antarctic Matters to prepare a draft of a comprehensive agreement with the U.S.A. on the joint efforts and research needed for preventing further radioactive contamination of the Arctic. This means that international projects will be strongly supported by the Russian government.

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**Session 2: Routes, Rates and Reactions**

Texts Accompanying Presentations

## Artificial Radioactivity, and Mixing in the Arctic

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### Abstract

Recently, we have become aware of the disposal of relatively large quantities of radioactive materials in the adjacent seas of the Arctic Ocean by the former Soviet Union. This includes thousands of barrels of apparently low level radioactive waste, some spent marine reactors (some containing fuel elements), a submarine, and effluent which enters the Arctic Ocean from rivers. In some respect the Arctic Ocean may have appeared to be an acceptable ocean dumping site for these materials. The sites are remote and away from population centers. There are no major fishing grounds near the dump sites. The sites are, to the best of our knowledge, within the coastal waters of the former Soviet Union. However, they are in waters over which the Soviet Union exercised careful control, releasing very little oceanographic data, and allowed only very limited access by foreign researchers. The question is: what are the risks associated with these disposals? Unfortunately, this question is very difficult to answer with the available information. The Arctic Ocean is a unique environment where we have very little long term experience with pollution.

The Arctic is a unique environment in many respects, which may make this apparent good disposal site questionable. The Arctic Ocean has a strong pycnocline separating the surface water from the deeper waters, and the vertical exchanges is restricted. The exchange of shelf waters with the basin waters has not been extensively studied. Tidal forcing is small throughout Arctic Ocean and adjacent seas. The limited kinetic energy input results in significantly lower turbulent and internal wave energy levels which are important in mixing processes.

Detailed knowledge of the Russian Arctic is limited in the West, however, it is believed that the Soviets had extensive knowledge about these waters. It is hoped that the Russians will be forthcoming with their knowledge of this environment, and share whatever data they hold which would help to quantify the risk.



## 1.0 Introduction

Recently, the Russians have reported that the former Soviet Union had dumped a significant quantity of radioactive materials in the shallow seas and rivers emptying into the Arctic Ocean (Yablokov, et. al., 1993). The Soviets have reported that the dump sites are on the continental shelf (some in bays) in depths between 20 and 300 m. We assume the liquid wastes, which are dumped in deeper ice free waters to the West of Nova Zemlya, are rapidly dispersed in the water column. If the liquid wastes are sufficiently dense or contain particulates then they may go directly to the bottom where they could accumulate. It is the higher level materials contained in drums and decommissioned reactors that are on the bottom that may pose the greater threat when they begin to leak. The materials might be dammed by the topography until a sporadic event would release them onto the surrounding shelf.

There are a number of factors which must be addressed if there is to be a proper assessment of the risk associated with the disposal of radioactive waste in the ocean and particularly in the Arctic regions. These factors are: (1) establishing the source term, (2) identifying the potential path ways, (3) determining the hazards to the ecosystem, and (4) assessing the available mitigation options and related costs. The source term involves the quantity and type of radionuclides, the concentrations and forms of materials dumped, the activity level, and the decay by-products. The paths in the Arctic include, air, water, ice, sediments and the food chain.

The Arctic is a unique environment and we have little experience with ocean dump sites in this region. Furthermore, it is difficult or inappropriate for us to try to transfer our experience from open ocean dump sites to an Arctic Ocean site. There are many aspects of the Arctic which must be considered in development of a risk assessment. These have been well documented in other papers at this conference and at the Workshop on Arctic Contamination (IARPC, 1993). The following is a list of some of these aspects:

- The Arctic Ocean is an enclosed sea with deep exchange only through the North Atlantic.
- The Arctic Ocean is centered around the North Pole and therefore the heat losses to the atmosphere are large.
- The central Arctic Ocean has a perennial cover of sea ice that is approximately 3 m thick on the average, but with pressure ridged ice which may be ten times this thickness.

- The ice cover insulates the ocean from exchanges of heat, moisture and momentum.
- The Arctic Ocean has a very strong pycnocline which further inhibits vertical mixing.
- A large and sudden influx of fresh water is added to the surface layer during the summer melt season (both due to ice melt and river runoff).
- Shelf processes (ice formation and movement, circulation and mixing, and ice-sediment interaction) are important but have not been studied extensively, at least in the West.
- In particular there is precious little environmental data from the dump sites.

All of the above are important issues in assessing the risk associated with the Soviet and now Russian dumping program in their coastal Arctic waters. Other contributors to this Conference and to the Anchorage Workshop have covered our background understanding of many of these issues in some detail. We want to focus on the issue of mixing processes which will play an important role in the transport and dispersion of material from the dump sites.

## 2.0 Mixing in the Arctic

In the open ocean mixing is driven by convective processes and shear. Air sea transfer, frontal and tidal phenomena play an important role. In the Arctic Ocean the geographic setting, ice cover and stratification create a unique mixing environment. We might divide the discussion up into basin scale, mesoscale and small scale processes. Smith (1990) provides an excellent summary of these processes in chapters by Carmack, Muench, and McPhee respectively. We will not summarize these further in this report. Instead we will focus on two important processes. First, is the exchange between the waters on the shelf and those in the deep basin. Secondly, the vertical mixing through the pycnocline. These processes are shown schematically in Figure 2.1.

During the period of ice growth the salt in the form of highly concentrated brine drains from the ice. Since the sea water is near its freezing point, where the density is not very dependent on temperature, the high concentration of salinity in the brine drives a convective overturn to the bottom. The cold, high salinity brine water accumulates on the bottom where gravity creates density driven cross shelf flows. Such cold, saline flows have been measured on the Canadian Continental Shelves (Milne, Personal Communication). The temperature minimum that is found in the Eurasian Basin just below the surface layer (100 m) has

been attributed to shelf derived brine waters. The cross shelf mixing of these waters with the slope waters may be augmented by topographically trapped flows such as is shown in Figure 2.2. Here the current is topographically controlled along the slope as it is advected along the edge of the Barents Sea. The mixing would be enhanced within these flows. Submarine canyons also are known to play a role in the off shelf movement of these waters (Aagaard and Roach, 1990). These may be augmented by pressure gradients that result in very strong down canyon flows. Mesoscale eddies may also be important in the cross shelf flux of water into the central basin and mixing in the pycnocline of the central basin.

Once a contaminant moves off the shelf it will be dispersed into the central Arctic. The Arctic has a very strong pycnocline so that vertical mixing may be inhibited. Indeed, measurements of internal wave and turbulent kinetic energy in the central Arctic have found that these mixing processes have significantly lower energy levels than are found in most open ocean measurements. Levine, et al, (1987) report that measurements in the Beaufort Sea of the total wave energy, integrated over the internal wave frequency band, were lower by a factor of 0.03 to 0.07 of energies observed in lower latitudes. Figure 2.3 shows the vertical wave number spectra of horizontal kinetic energy in the Arctic compared to other ocean regions. The reduced energy levels in the central Arctic are evident. However, it should be noted that the energy levels on the Yermak Plateau are higher than the open ocean measurements collected in the Sargasso Sea and the North Pacific. It appears that this is a region of enhanced mixing.

Padman and Dillon (1987), investigating vertical heat fluxes through the thermohaline staircase between 320 and 430 m in the Beaufort Sea, found kinetic energy dissipation rates in some cases below the noise floor of their airfoil probes. In order to explain the basin averages vertical heat fluxes from the Atlantic layer one needs to identify mechanisms other than vertical diffusion of heat. It has been suggested that the Mesoscale eddies (Aagaard, 1993) or benthic stirring where the layer intersects the continental shelf and slope are the main sources of mixing. Figure 2.4 shows representative vertical profiles of horizontal velocity profiles collected during AIWEX in the Beaufort Sea when a mesoscale eddy was present and absent. The Arctic profiles are compared to one collected in the California Current. The low energy levels in the Beaufort when no mesoscale eddies are present is apparent.

Figure 2.2 shows how the currents flow along the edge of the upper continental slope in the southern Eurasian Basin. This would be a region of enhanced mixing. Kowalik and Proshutinsky (1993) have shown that the diurnal tides, while

low in amplitude, excite resonant-like interactions on the continental shelf, such that currents are enhanced. In these regions the flux of energy from the diurnal tides is amplified by shelf wave interaction, and afterwards the tidal energy is strongly dissipated in the same regions.

### 3.0 Summary and Conclusions

The Arctic Ocean is unique in many respects which may make this apparent good disposal site questionable. The Arctic Ocean has a strong pycnocline separating the surface water from the Atlantic layer, and the vertical exchange is restricted by this fact. Momentum exchange from the atmosphere to the ocean is restricted by the stratification and the presence of the Arctic ice pack. Tidal forcing is small throughout most of the Region. Fresh water input is restricted except during the summer melt season. The limited amount of kinetic energy input results in significantly lower energy levels in turbulence and internal wave spectra. Mesoscale eddies, shelf processes, and boundary currents may play an important role in the dispersion of contaminants from the shelf area into the central Arctic. Detailed knowledge of Russian Arctic waters is limited in the West, however, it is believed that they have extensive knowledge about these waters. It is hoped that whatever data and information is available that bears on the question of the fate of contaminants dumped in the Arctic, and objective risk assessment, will be made available by all parties.

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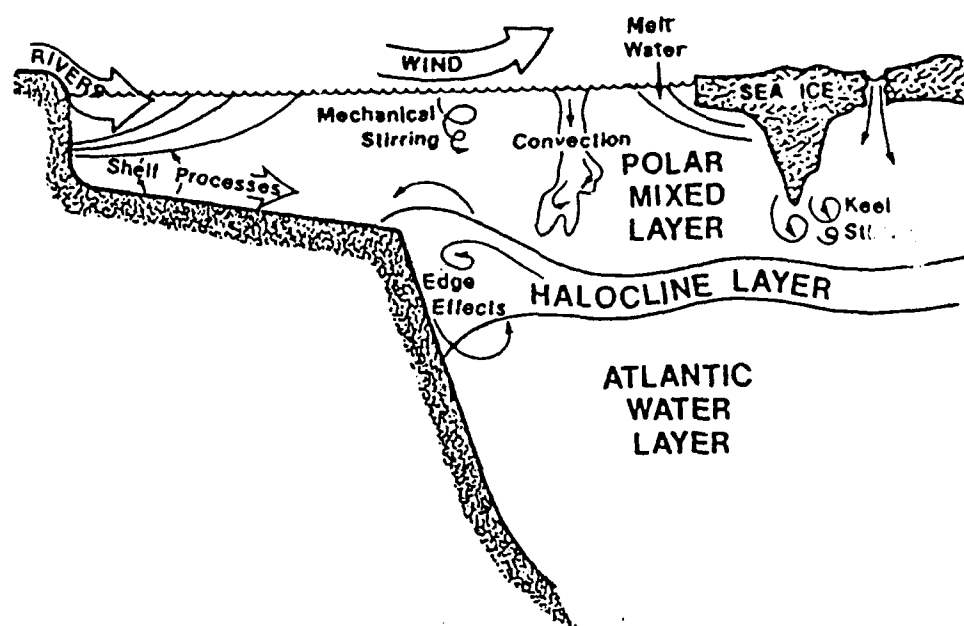


Figure 2.1 Exchange and internal mixing processes in the Arctic Ocean.

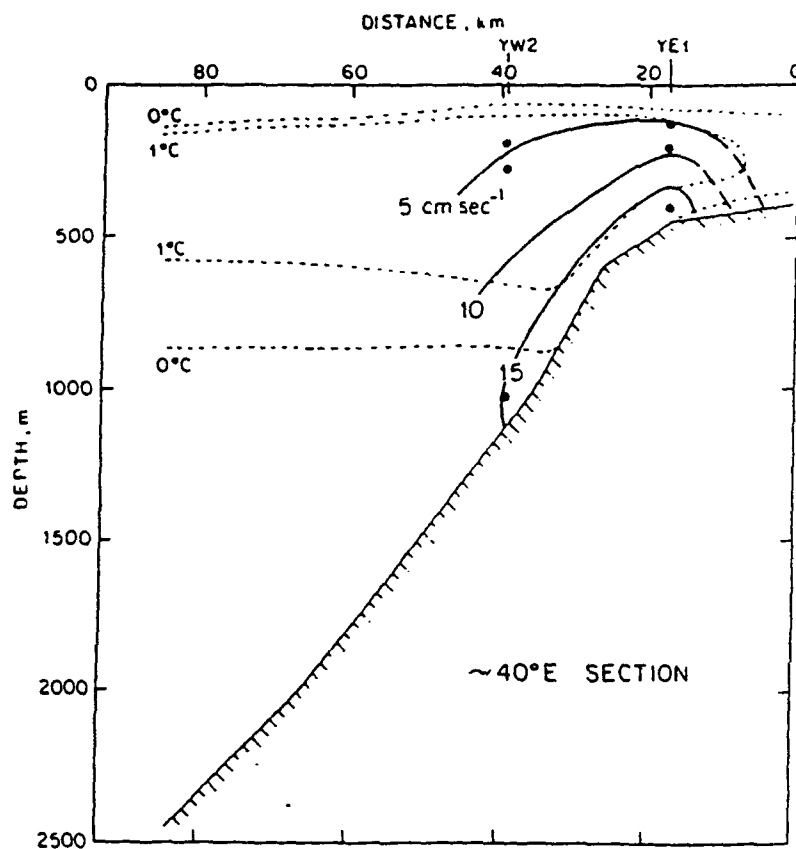


Figure 2.2 Composite mean velocity section across the southern Eurasian Basin slope near 40°E from July-September 1980. The current is directed into the figure. (From Aagaard, 1993)

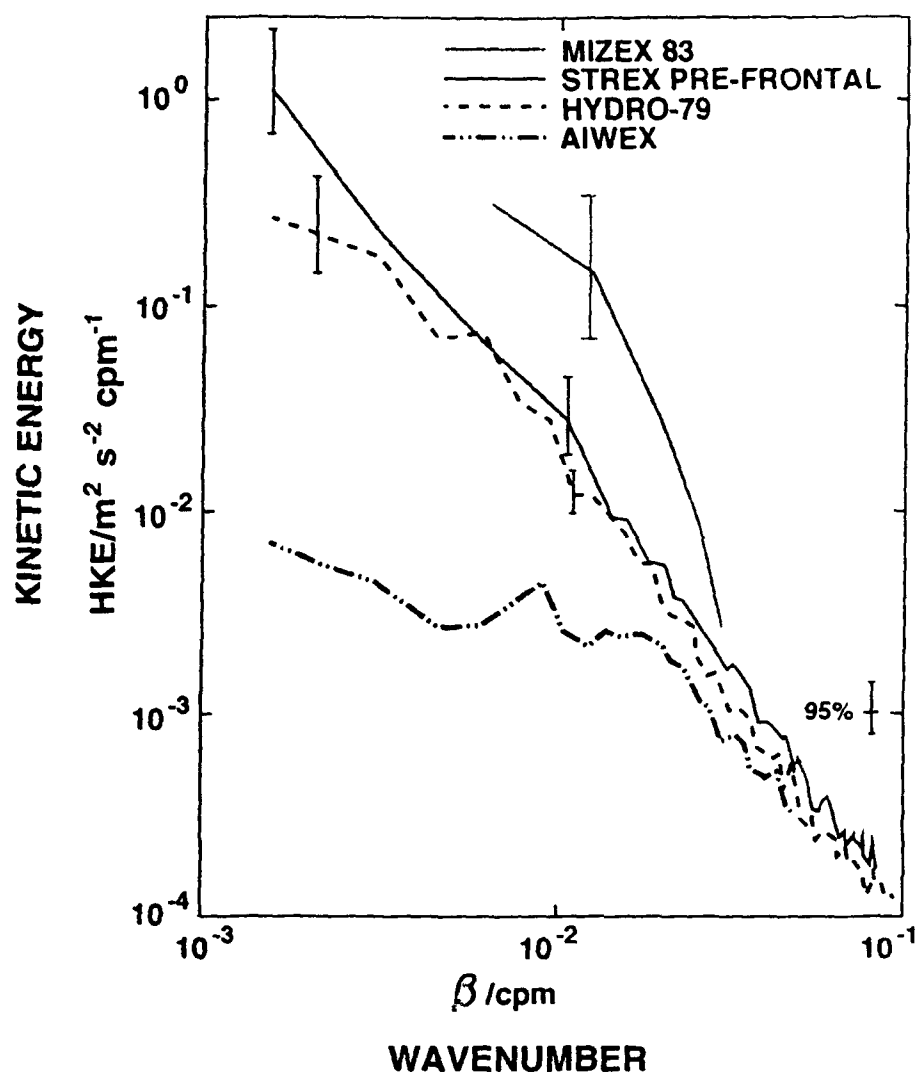


Figure 2.3 Vertical wave number spectra of horizontal kinetic energy from the Arctic Basin (AIWEX), the Sargasso Sea (HYDRO-79), North Pacific (STREX), and the Yermak Plateau (MIZEX 83).



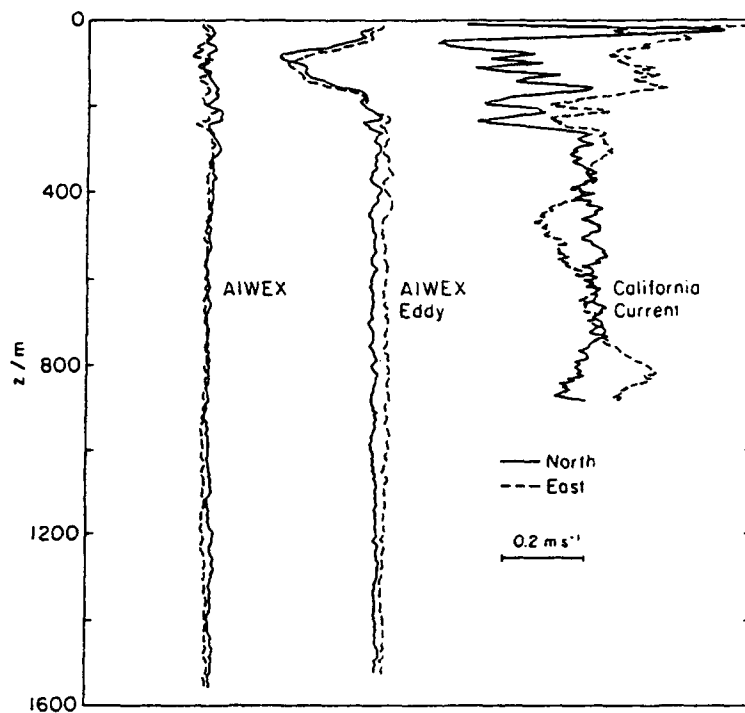


Figure 2.4 Representative velocity profiles taken during AIWEX with and without the presence of an eddy, and off California ( $31^{\circ}\text{N}$ ,  $121^{\circ}\text{W}$  on October 9, 1992).

First International Conference  
Radioactivity & Environmental Security in the Oceans:  
New Research and Policy Priorities in the Arctic and North  
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**Radioactivity in the Barents Sea, past and present status,  
and its impact on fisheries.**

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**Summary**

During the atmospheric nuclear bomb test at the end of the fifties and in the beginning of the sixties the Institute of marine research, IMR, monitored the radioactive contamination in commercial landed fish from the Barents Sea. There were indications of an immediate response in uptake of radionuclides depending on the time of the year, probably due the food situation for the fish. There was also indications of species dependant uptake of radionuclides in fish. Even during the most intensive test period with fall-out directly to the Barents Sea the total beta-activity never exceeded 80 Bq pr kg fish.

The last years media focus on potential radioactive contamination in the Barents Sea have necessitated an establishment of a quite extensive monitoring program, both in fish, water and sediments. The area of interest extends from the site of the sunken former Soviet submarine, "Komsomolets" in the west to Novaya Zemlya in the east of the Barents Sea.

There is at present no significant indication of elevated contamination due to the dumping of radioactive waste by the former Soviet Union. The only present serious concern to the fisheries is the enormous focus from the media on the sunken submarine, "Komsomolets", and the dumped radioactive material in the Kara Sea.

**The Barents Sea**

The Barents Sea, Fig. 1, is a shallow sea with an average depth of 230 m, and covers the area between 70°N and 80° N and from the west at the rise of the slope from the depth of more than 2500 m in the Norwegian Sea to the coast of Novaya Zemlya in the east. With an area of 1,4 million km<sup>2</sup>, the Barents Sea represents only about 7% of the total areas of the Arctic ocean. As an extreme maximum during winter and spring, as much as 75% of the surface can be covered by ice. The annual variation is however



Fig. 1. Bathymetric map of the Barents Sea (Loeng, 1989).

considerably. Most of the ice melts during the summer and creates by this process a special hydrographic regime with a rapid developing ice-edge phytoplankton bloom (Skjoldal and Rey, 1989, Sakshaug and Skjoldal, 1989). This highly productive zone follows the ice-edge retreating northwards during the melting.

The inflow of warm, nutrient rich Atlantic water to the Barents Sea is

another essential factor influencing the conditions for biological production in the area. The Barents Sea is, however, characterized by large fluctuations in the inflow as well as large seasonal and inter-annual variability in the ice cover.

Representing only about 0,4 % of the total surface of the world oceans, this area produces about 4 % of the total world fish catches. The Barents Sea ecosystem contains some of the world largest fish stocks like the capelin, the Northeast Atlantic cod and partly the Norwegian spring spawning herring. There are strong interactions between these stocks, and variations in the year-class strength have a marked influence on other components of the ecosystem (Hamre, 1991).

In addition to the direct harvest of the area the value as feeding grounds for fish populations harvested further south on the Norwegian shelf is considerable. The Norwegian shelf area from 62° N and northwards is spawning grounds for the most important fish populations of the Northeast Atlantic. Fish-egg and -larvae are transported via the Norwegian Coastal Current to the Barents Sea.

The annual catches of fish from the Barents Sea have, during the last forty years, been in the order of 2,0 - 3,5 million tonnes. There have been a considerably variation in the catches due to both overfishing and changing environmental conditions. This variability is illustrated in figs 2 and 3, which summarises the last ten years landings of capelin, cod and haddock.

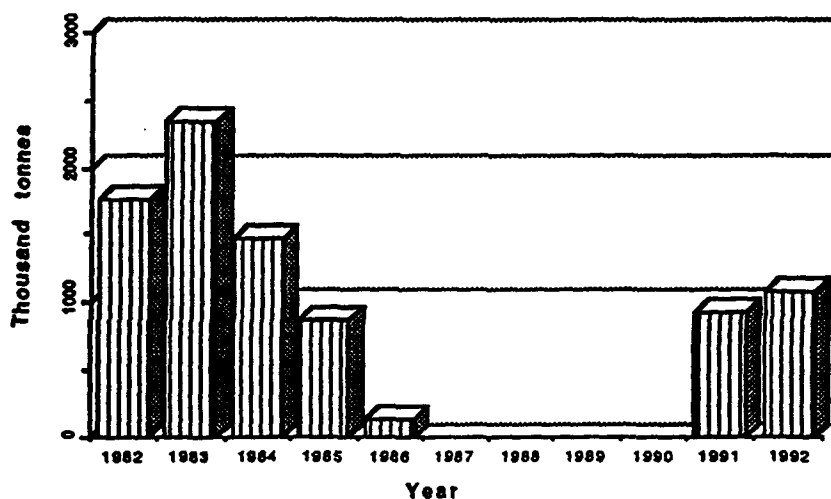


Fig. 2. Yearly catches of capelin (*Mallotus viosus*) in the Barents Sea,

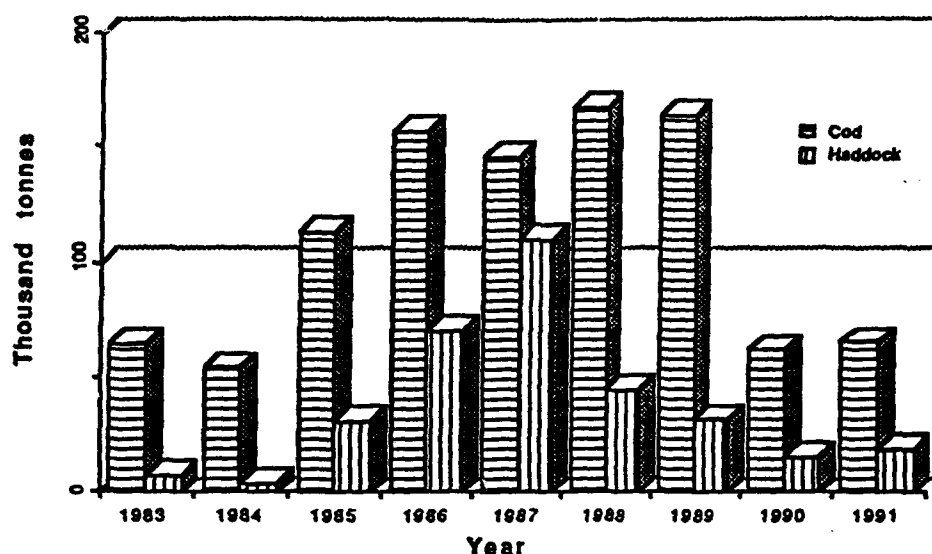


Fig. 3. Yearly catches of cod (*Gadus morhua*) and haddock (*Melanogrammus aeglefinus*) in the Barents Sea.

### Radioactive contamination

#### a) Observations in the sixties

In the late nineteen-fifties rumours about radioactive contamination of the marine resources of the Barents Sea began to affect the Norwegian fishing industry. The Institute of Marine Research, IMR, Bergen, as an advisory research institute under the Norwegian Ministry of Fisheries, started then, in 1958, a monitoring program on radioactivity in fish meal produced by a fish-fillet factory in Hammerfest. The fish meal was processed of left-over from the fillet fabrication, i.e. skin, bones and other non edible fish tissues. The samples represented an average of the catches from the various fishing areas, and the measurement was aimed to detect possible radioactive contamination in commercial landed fish.

The intense Soviet nuclear bomb tests at Novaya Zemlya in 1961 initiated, in addition to the ongoing monitoring of fish-meal, regular measurements on the most important fish species, i.e. cod and haddock, on an individual basis. A summary of the results of these measurements is presented in fig. 4, (Føyn, 1991). The values at the time of measurement were reported as

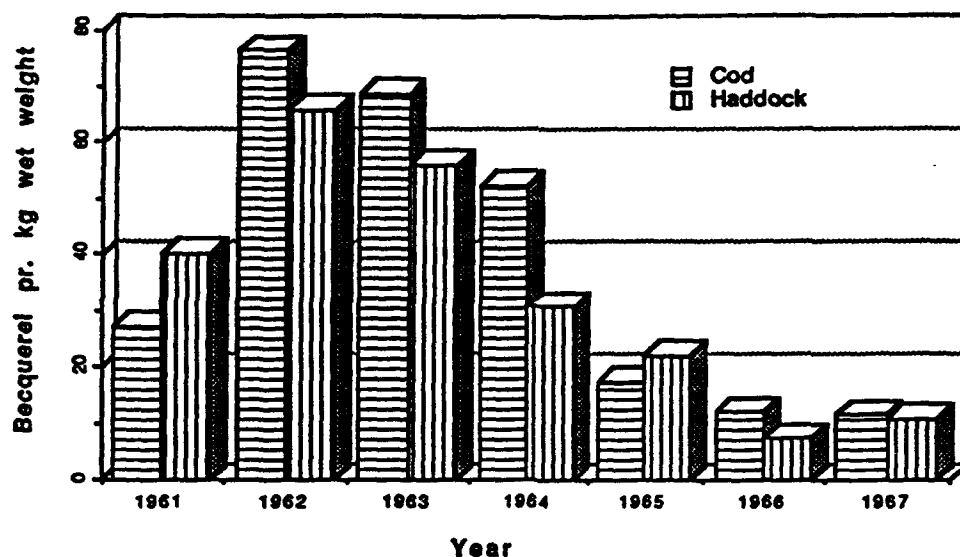


Fig. 4. Yearly mean values of total anthropogenic beta-activity in the edible part of cod (*Gadus morhua*) and haddock (*Melanogrammus aeglefinus*) from the Barents Sea in the period from 1961 through 1967 (Føyn, 1991).

total beta-activity minus the activity of potassium - 40, (Berge pers com.), and the potassium-40 values were calculated on the basis of a constant isotope ratio after determination of total potassium content in aliquots of the samples by the use of flame spectrophotometry, 1g K giving 1.740 dpm beta-activity.

Although the monitoring was not specific regarding determination of radionuclides, the results gave a sufficiently good basis for a continuous assessment of radioactive contamination of fish and fish products from the Barents Sea. The collected data was used in an advisory contexts and unfortunately very few of the results were published outside internal notes. The observed contamination varied from an average maximum value of close to 80 becquerel pr kilogramme wet weight (Bq kg<sup>-1</sup> w.w.) to below 10 Bq kg<sup>-1</sup> w.w. in 1968, a value believed to be at the anticipated background level.

At the time of closure of the monitoring programme in 1968 it was believed (hoped?) that the problem of radioactive contamination in the fish resources of the Barents Sea was a thing of the past. The Chernobyl accident and more recent events as the wreckage and sinking of the

former Soviet nuclear submarine "Komsomolets" and the documentation by the Russian government (Annon, 1993) of the problem of dumped radioactive material at the east coast of Novaya Zemlya, have proved different.

The measurements undertaken during the monitoring programme increased our knowledge of the behaviour of direct fallout to the ocean. The nuclear bomb tests conducted in the eastern Barents Sea created a considerable direct contamination of this sea area. An example of such direct contamination was experienced by one of our research vessels working close to the test area during a detonation in the late fifties, measurements undertaken more than one month after the visit to the actual area and after several wash-down of the whole ship, showed considerably activity especially in ropes and tarpaulins.

The collected data from the monitoring program showed clearly differences in contamination between the fish-meal samples and the individual samples. (Sampling frequency was in the most intense period twice a week.) While contamination was clearly demonstrated in the individual samples, muscle tissue, the fish-meal which consisted of skin and bone and other non edible parts as the gills, where water flushes through constantly, showed less evidence of contamination. This indicate that the uptake of radionuclides was through the food. The monitoring showed also an "immediate" response of higher contamination following a bomb test.

There was also a pronounced seasonal difference in the radioactive contamination of the fish. In the summer situation with high secondary production following the phytoplankton bloom the radioactive contamination was also at its highest. Indicating that the bio-concentration was through phytoplankton to zooplankton. During the highly productive summer season the various fish species may shift their diet to smaller species.

Berge (pers. com.) observed variation in the composition of radionuclides from one contamination period to another. Cod samples from October 1961 were dominated by rather short-lived radionuclides while in cod samples from October 1962, although a pronounced presence of short-lived, the long-lived radionuclides dominated. Iodine-131, manganese-54, zinc-65, ruthenium-106 and cesium-137 contributed to the contamination.

Another observation made by Berge was the species dependent uptake of strontium-90 in fish bones. During the maximum radioactive

contamination period in the summer of 1962 strontium-90 was measured in cod, haddock and spotted catfish. While significant levels of strontium-90 was detected in the samples of haddock, only insignificant levels were determined in cod and spotted catfish.

The monitoring program was closed in 1968 when the situation concerning radioactive contamination of the marine resources of the Barents Sea, was found to be at the anticipated background level.

#### **b) Present observations**

When the Chernobyl accident happened almost two decades had passed without any systematic monitoring of the radioactivity in the Barents Sea. The accident initiated new activity in this field at the Institute of Marine Research, IMR. New instrumentation was provided through a special governmental funding and by 1990 we started the sampling of sediments and water in the Barents Sea for, in the first hand for determination of radiocesium.

#### **Sediments**

Figure 5 presents the results of the measurement of cesium-137 in sediment as becquerel per square meter from the upper 1 cm. Highest values, amounting to about 150 Bq m<sup>2</sup>, are found in the eastern part of the Barents Sea. As can be seen from the figure the highest values are also found in the vicinity of areas where there have been discharges of low-level liquid waste (Annon, 1993).

#### **Biota**

Some fish samples (5) from the Barents Sea have been measured by the Norwegian Radiation Authority, values between 1,6 - 3,3 Bq kg<sup>-1</sup> w.w. for <sup>134</sup>Cs + <sup>137</sup>Cs were found (Selnæs pers. com. 1993). The Directorate of Fisheries is monitoring fish from commercial landings and so far no values have exceeded their detection limit of 20 Bq kg<sup>-1</sup> radiocesium.

The Russian fisheries institute, PINRO, in Murmansk, measured radioactive contamination in seaweed from the eastern part of the Barents Sea in 1991, they reported values from 4 - 10 Bq kg<sup>-1</sup> dry weight. Earlier measurements reported, 1980 - 1983, from the western Barents Sea and the Norwegian coast are within the same range.

#### **Water**

Our measurements in surface samples collected in 1991 showed values of cesium-137 between 5 and 15 Bq m<sup>-3</sup>. These values are within the same range as the values determined on board the Russian research vessel "Akademik Boris Petrov" on a cruise along the Norwegian coast and in



the Barents Sea in June 1991. The results from these measurements showed a distinct reduction in cesium-137 levels in the surface water as the ship

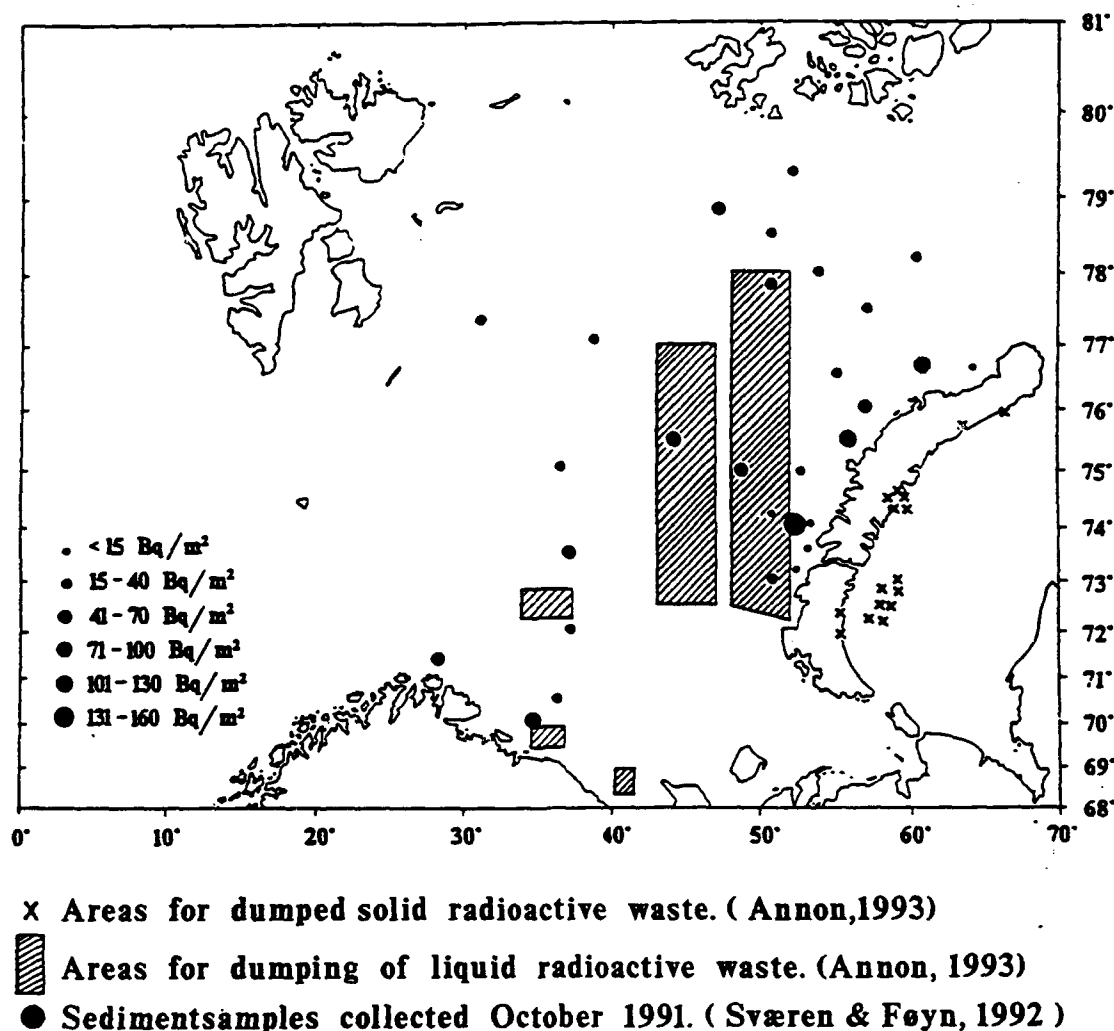


Fig. 5. Cesium-137 values in sedimentsamples from the Barents Sea and areas and sites of the former Soviet dumping of radioactive waste.

moved northwards, from about 50 Bq m<sup>-3</sup> in the Skagerrak at the southern border of Norway, to values around 10 Bq m<sup>-3</sup> in the Barents Sea.

The results from the on board preliminary measurements of cesium-137 during the Norwegian - Russian expedition to the Kara Sea, east of Novaya Zemlya, showed surface water values around 5 Bq m<sup>-3</sup> (Føyn and Semenov, 1992).

### **The "Komsomolets" - situation**

The media focus on the sunken former Soviet submarine is continuing in an astonishing force. The submarine has found its rest at the depth of 1658 m in the position 73° 43,49' N and 13° 15,96' E. The water masses at this depth are more or less closed off from the upper layers due to the hydrographic conditions. Any release from the wreck of radionuclides will slowly be mixed with the deep water masses in the Norwegian Sea, This water will only surface to biological active layers in some hundred years, and then in the southern part of the Atlantic Ocean. The enormous water masses available for dilution secure the potential radioactive contamination in the ocean due to the "Komsomolets" to be insignificant.

The potential harm to marine resources from the release of radionuclides from the "Komsomolets" presented in Russian documents (Annon, 1993) is far from realistic. They estimate an economic loss of 2,5 billion rubles (1991 values) for the fisheries.

In this context we have reason to underline the fact that there is now fisheries in the area and as long as the submarine is located at the bottom and an eventually release of radionuclides will take place at that depth no fish resources is ever likely to be contaminated from this source.

The Russian report (Annon, 1993) points to the release of plutonium from the two nuclear warheads in the wreck. There is however some experience already of the behaviour of plutonium from nuclear warheads in the marine environment. After the wreckage of an U.S. aircraft with nuclear bombs close to Thule, West-Greenland, 1968, monitoring of plutonium migration in the marine sediment showed a movement outwards from the source of about 400 m pr year.

### **Concluding remarks**

The fish resources as such of the Barents Sea have not yet been affected by anthropogenic radioactivity. Neither during the nuclear bomb tests in the fifties and sixties, nor during recent years due to accidental releases. The fisheries may however, be dramatically affected by the fact that the focus of the media on radioactive contamination frightens people from eating fish. Television programmes with "nice" pictures of nuclear explosions combined with pictures of fishing activities and further a dish with fish with a question-mark of the edibility of the food are not

especially in favour for the fisherman and the fishing industry.

The problem of radioactive contamination in the marine environment is to this day far less than the problem of contamination by organic micro pollutants. Sætre et. al., 1992, in summarizing the situation in the Barents Sea, points to the fact that persistent organic pollutants as PCBs, are found in all marine species of the area. These pollutants are concentrated through the food-web and create a potential threat to animals as seal and polar bear.

For the concern of the radioactive contamination of the Barents Sea we will continue the monitoring,

- a) to detect possible changes in the presence of radionuclides in Norwegian waters, and to develop a base for assessing, in a readiness situation, the impact of potential accidental releases, and
- b) to document the potential of contamination both in marine fish and in the marine environment for the purpose of avoiding speculations about the quality of fish products from Norwegian waters.

#### **Acknowledgment.**

*The documents dealing with the problem of radioactive contamination in the fifties and sixties, both hand written and data sets not published have been made available from the personal files of former research director at the IMR, Dr. Grim Berge. I would like to use this opportunity to thank him for valuable information and inspiring discussions about marine radioactivity.*

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## BEHAVIOR OF ARTIFICIAL RADIONUCLIDES IN SEAWATER

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Over the four decades of nuclear materials production, vast quantities of long lived artificial radionuclides have accumulated in the biosphere. The greater part of their world inventory is located at storage facilities on the continents, although some have been released in the environment, particularly into the oceans.

The total amount and relationship among the masses of radionuclides in seawater is dictated by the type of source, time of release in the environment, and the physical, chemical and biological processes.

### 1. Sources of radionuclides in sea water

Among the main sources of radionuclides in the seas and oceans we can indicate the following:

- \* Radioactive waste disposal into the seas;
- \* Marine accidents involving nuclear-powered submarines;
- \* Base regions for atomic submarines and other war  
- and civil ships with nuclear reactors;
- \* Explosions from nuclear weapons testing;
- \* River waters which transfer the radionuclides from:
  - radiochemical plants of military industrial complexes  
for the production of nuclear weapon components;
  - areas for setting off atomic explosions for peaceful uses;
  - nuclear power stations and temporary nuclear waste  
repositories resulting from their activity;

Nuclear weapons testing has been the first major single source of radioactive contamination of the oceans [Noshkin and Bowen, 1973]. The former Soviet Union conducted nearly all of its high-yield tests at Novaya Zemlya, an island in the Arctic Ocean.

The dumping of radioactive waste in seas and oceans began in 1946 when the United States sank containers of radioactive waste in the Pacific Ocean some 80 km off the California coast. Over the decades 12 countries including Belgium, Great Britain, France, Germany, Italy, Japan, the Netherlands, New Zealand, South Korea,

Sweden, Switzerland and the United States have declared that they dumped radioactive wastes at more than 50 sites in the northern part of the Atlantic and Pacific Ocean [Prandle and Beechey, 1991; Smith et al., 1990]. The International Atomic Energy Agency has estimated that these nations from 1946 to 1982 dumped a total of 1.7 million curies of radioactive refuse into the world oceans [Calmet and Sjoebloom, 1992].

Recently, a report of the governmental commission headed by Dr. A.V. Yablokov [Yablokov et al., 1993] states that the former Soviet Union dumped 2.5 million curies of radioactive wastes, including 18 nuclear reactors from submarines and an icebreaker. Sixteen of these reactors were cast into the shallow waters of the Kara Sea, six of them heavy with radioactive fuel.

## 2. Peculiarities of the behavior of radionuclides in sea water

In theory, it is possible for the ocean to dilute liquid radioactive wastes rendering them essentially harmless. However, localized releases of radionuclides in high concentration can do real damage when picked up by marine life. Fortunately, the Kara Sea is frozen nine months of the year and has little biological activity. The fishing grounds of the Barents, White and Norwegian seas lie hundreds of kilometers away. Such distances limit but do not eliminate the physical and biological migration of radionuclides from dumped radioactive wastes.

For many chemical reactions involving radionuclides chemical equilibrium is not achieved because of very slow rates. Thus, the predominant species of plutonium in marine water from the thermodynamic point of view is Pu(VI). But, as Morse and Choppin [1986] show, plutonium should exist as Pu(V). Organic matter plays an important role in the reduction of the Pu(V) aqueous species. The reduction of the Pu(VI) aqueous species in Pu(V) takes place in a short time, but Pu(V) in Pu(IV) considerably inhibits.

In studies of artificial radionuclides behavior in marine waters it is important to keep in mind that in the majority of cases their concentrations are very insignificant and the formation of own phases can not take place. It is possible only in the case of the spent fuel of nuclear reactors of sinking submarines interacting with sea water. In this case the thermodynamic data for radionuclide bearing solid compounds, their solid solutions, and kinetic information on rates of chemical reactions and their dissolution (and deposition) in marine waters is important.

The results of many investigations on radionuclide contamination of the seas and oceans [Aston and Duursma, 1973; Beasley et al., 1982; Buchholtz ten Brink, 1987; Erlenkeuser and Balzer, 1988; Higgo and Rees, 1986; Lieser et al., 1986;] show that water-sediment interaction appears to be one of the main factors controlling the radionuclide behavior in this case. It may be concluded that their behavior in bottom sediments is controlled by the following processes:

- \* sorption on the sediments and soil particles,
- \* radionuclide diffusion in sediment pore water,
- \* radioactive decay of radionuclides in bottom sediments.

The effect of these processes on the radionuclide concentration in sea water should be separately considered for different regions of the bottom sediments characterized by different mineral composition and levels of initial radionuclide contamination. For the chosen region of bottom, the sediments might be considered as a uniform media which could be characterized by the distribution ratios for different radionuclides (quantifies the partitioning between solid and solution due to sorption), the porosity, the sediment grain density, the tortuosity.

Another important factor controlling artificial radionuclide behavior is the biological control, which influences the vertical flux of radionuclides in marine waters and their deposition in sediments [Fowler, 1989].

### 3. Thermodynamic data for physico-chemical analysis of behavior of the radionuclides in sea water

The long time predictions of radiological consequences in the case of entry of radionuclides (actinides and radioactive isotopes of Cs, Sr, Pu, Ru and others) into the environment require the numeric description of the chemical substances' behavior in natural systems.

For numerical computer modelling of natural systems and processes various recommended values of physico-chemical data such as thermodynamic data on rock-forming and weathering minerals, aqueous species, solubility and equilibrium data, oxidation-reduction potentials and other physico-chemical data are needed. They have to be obtained by the expert analysis of numerous experimental data for substances and chemical reactions which characterize the natural processes. The main aim of such analysis is



to obtain the recommended values with the uncertainty which determines the quality of experimental information.

The precision of the predictions of the behavior of chemical elements in the many natural and technological systems has been constrained by the uncertainties of thermodynamic data. Large discrepancies between different data exist in many cases. For example, the Gibbs energies of formation and entropies of the all uranium compounds in aqueous solutions as well as some crystalline substances are based on the thermodynamic values accepted for the aqueous  $\text{UO}_2^{+2}$  ion, which is, therefore, a key value for the chemical thermodynamics of uranium. Also many estimated values of standard entropies of actinide compounds are based on corresponding values of uranium bearing species in aqueous solutions.

Recently P.A.G. O'Hare et al.(1988) have revealed appreciable differences (nearly an order of magnitude) in the values of the solubility product of  $\text{UO}_2(\text{OH})_2 \cdot \text{H}_2\text{O}$  (Cr, Schoepite). One is calculated from thermochemical data and another is obtained from the results of solubility measurements. Later Bruno and Sandino (1989) supported this conclusion by their new experimental measurements of the solubility product of Schoepite. The cause of the divergence may be an error in the thermodynamic values of uranyl hydroxide [ $\text{UO}_2(\text{OH})_2$ ] or those of the uranyl ion. As has been shown by analysis of the experimental data on thermochemistry and equilibrium measurements in the  $\text{UO}_3 - \text{H}_2\text{O}$  system [Khodakovsky, 1992], thermodynamic values of uranyl hydroxide,  $\text{UO}_2(\text{OH})_2 \cdot \text{H}_2\text{O}$ , agree well with those for other solids in the system  $\text{UO}_3 - \text{H}_2\text{O}$ .

Since the CODATA recommended formation enthalpy of  $\text{UO}_2^{+2}$  - ion does not raise doubts, there is a possibility that the CODATA recommended standard entropy value of uranyl ion has a systematic error. That assumption is confirmed by calculations of standard entropies of uranyl ion, using thermodynamic information for the reactions in the systems  $\text{UO}_3 - \text{CO}_2 - \text{H}_2\text{O}$  and  $\text{UO}_3 - \text{SO}_3 - \text{H}_2\text{O}$  [Khodakovsky, 1992].

Reliable data on solubilities of inorganic compounds of radionuclides (actinides and radioactive isotopes of Cs, Sr, Pu, Ru and others) are extremely needed for prediction of the behavior of those elements in the case when the spent fuel of nuclear reactors of sinking submarines interacts with seawater.

In connection with that, all available published results of the experimental investigations of the solubility, hydrolysis and complex formation processes occurring in aqueous actinide and Cs and Sr-

bearing systems have been collected and analyzed within the framework of an update for the computer database "DiaNIK" which has been developed at the Vernadsky Institute of Geochemistry and Analytical Chemistry and Institute of Experimental Mineralogy of the Russian Academy of Sciences.

"DiaNIK" is an interactive research information system of chemical thermodynamic data for minerals and mineral-forming substances. The structure of thermodynamic database "DiaNIK" is based on the principles, developed by the CODATA International Group on Geothermodynamic Data for working up thermodynamic information. The accepted structure allows performance of the consistency procedure both for the individual substance or the individual chemical reaction (local consistency), and for the chemical systems, containing a set of substances and reactions (global consistency). The "DiaNIK" system core supports the work of separate blocks, which can provide different opportunities and alternatives for the user. At present successive local consistency procedures are used for treatment of thermodynamic data in the database.

A substantial portion of the thermodynamic values recommended by "DiaNIK" experts for the substances in User Version 3.1 resulted from the analyses based on review of the literature data, making use of original algorithms of the Expert Version 3.2.

The validity of experimental data on the radionuclide compounds solubility and the stability constants of aqueous complexes is controlled by a series of factors. Among those the following ones should be primarily considered for solubility data:

- an achievement of equilibrium in the studied system;
- existence of ions in the system, which are capable of complex formation with the cation of the solids);
- existence of reducing or oxidizing species in the system (if containing solids with an element of variable valence);
- method of separation of solid from the solution before the analysis;
- the dispersion degree of the solid
- radioactivity influence (radiolysis, metamict phenomena, etc.).

The results of the critical evaluations of the thermodynamic data for the actinide aqueous inorganic complexes, which were published before 1989, have to be obtained from [Fuger, J.,

Khodakovsky, I.L., Sergeyeva, E.I. and others, 1992], and for uranium complexes also from [Grenthe et al., 1992]. This data in some cases was corrected by "DiaNIK" experts using the results of recent experimental determinations. The recommended values for stability constants of some aqueous complexes of U, Np, Pu, Am, Sr and Cs at infinite dilution are presented in the table.

TABLE  
Recommended Values for Stability Constants  
of Aqueous Complexes at 298.15 K and 1 bar

-----		
% *f(o logK(0)       +/-		
-----		
URANIUM		
$U^{+4} + H_2O(l) = UOH^{+3} + H^+$	-0.30	0.2
$U^{+4} + 3H_2O(l) = U(OH)^{+3} + 3H^+$	-1.1	0.1
$U^{+4} + 4H_2O(l) = U(OH)_4 + 4H^+$	-5.4	0.2
$UO_2^{+2} + H_2O(l) = UO_2(OH)^+ + H^+$	-5.79	0.10
$UO_2OH^+ + H_2O(l) = UO_2(OH)_2 + H^+$	-7.24	0.2
$UO_2^{+2} + 3H_2O(l) = UO_2(OH)^{-3} + 3H^+$	-20.2	0.1
$2UO_2^{+2} + H_2O(l) = (UO_2)_2(OH)^{+3} + H^+$	-4.09	0.15
$2UO_2^{+2} + 2H_2O(l) = (UO_2)_2(OH)_2^{+2} + 2H^+$	-5.51	0.04
$3UO_2^{+2} + 5H_2O(l) = (UO_2)_3(OH)^{5+} + 5H^+$	-15.41	0.10
$3UO_2^{+2} + 7H_2O(l) = (UO_2)_3(OH)^{7-} + 7H^+$	-3.1	
$U^{+4} + HSO_4^- = USO_4^{+2} + H^+$	6.4	0.5
$U^{+4} + 2HSO_4^- = U(SO_4)_2 + 2H^+$	7.6	0.7
$UO_2^{+2} + SO_4^{-2} = UO_2SO_4(ao)$	3.36	0.20
$UO_2^{+2} + 2SO_4^{-2} = UO_2(SO_4)_2^{-2}$	4.1	0.2
$UO_2^{+2} + 3SO_4^{-2} = UO_2(SO_4)_3^{-4}$	3.40	0.30
$UO_2^{+2} + CO_3^{-2} = UO_2CO_3$	10.0	0.2
$UO_2^{+2} + 2CO_3^{-2} = UO_2(CO_3)_2^{-2}$	17.0	0.4
$UO_2^{+2} + 3CO_3^{-2} = UO_2(CO_3)_3^{-4}$	21.6	0.1
$3UO_2^{+2} + 6CO_3^{-2} = (UO_2)_3(CO_3)_6^{-6}$	53.4	0.8

## NEPTUNIUM

$\text{Np}^{+4} + \text{HSO}_4^- = \text{NpSO}_4^{+2} + \text{H}^+$	5.4	0.7
$\text{NpSO}_4^{+2} + \text{HSO}_4^- = \text{Np}(\text{SO}_4)_2 + \text{H}^+$	7.6	0.9
$\text{NpO}_2^+ + \text{SO}_4^{-2} = \text{NpO}_2\text{SO}_4$	0.6	0.2
$\text{NpO}_2^{+2} + \text{SO}_4^{-2} = \text{NpO}_2\text{SO}_4$	3.4	0.3
$\text{NpO}_2^+(\text{ao}) + \text{CO}_3^{-2} = \text{NpO}_2\text{CO}_3^-$	4.9	0.3
$\text{NpO}_2^+ + 2\text{CO}_3^{-2} = \text{NpO}_2(\text{CO}_3)_2^{-3}$	7.05	0.20

## PLUTONIUM

$\text{Pu}^{+3} + \text{SO}_4^{-2} = \text{PuSO}_4^+$	4.5	0.5
$\text{Pu}^{+3} + 2\text{SO}_4^{-2} = \text{Pu}(\text{SO}_4)_2^-$	6.7	0.6
$\text{Pu}^{+4} + \text{HSO}_4^- = \text{PuSO}_4^{+2} + \text{H}^+$	5.5	0.5
$\text{Pu}^{+4} + 2\text{HSO}_4^- = \text{Pu}(\text{SO}_4)_2 + 2\text{H}^+$	7.7	0.7

## AMERICIUM

$\text{Am}^{+3} + \text{SO}_4^{-2} = \text{AmSO}_4^+$	4.2	0.4
$\text{Am}^{+3} + 2\text{SO}_4^{-2} = \text{Am}(\text{SO}_4)_2^-$	6.1	0.5

## STRONTIUM

$\text{Sr}^{+2} + \text{OH}^- = \text{Sr}(\text{OH})^+$	0.89	0.10
$\text{Sr}^{+2} + \text{NO}_3^- = \text{SrNO}_3^+$	0.72	0.10
$\text{Sr}^{+2} + 2\text{NO}_3^- = \text{Sr}(\text{NO}_3)_2$	0.79	0.08
$\text{Sr}^{+2} + 3\text{NO}_3^- = \text{Sr}(\text{NO}_3)_3^-$	0.50	0.20
$\text{Sr}^{+2} + \text{JO}_3^- = \text{SrJO}_3^+$	1.00	0.05
$\text{Sr}^{+2} + \text{Cl}^- = \text{SrCl}^+$	0.57	0.20
$\text{Sr}^{+2} + \text{HPO}_4^{-2} = \text{SrHPO}_4$	2.40	0.20
$\text{Sr}^{+2} + \text{H}_2\text{PO}_4^- = \text{SrH}_2\text{PO}_4^+$	0.73	0.20
$\text{Sr}^{+2} + \text{PO}_4^{-3} = \text{SrPO}_4^-$	5.7	0.2
$\text{Sr}^{+2} + \text{P}_3\text{O}_9^{-3} = \text{SrP}_3\text{O}_9^-$	3.5	0.2
$\text{Sr}^{+2} + \text{P}_4\text{O}_{12}^{-4} = \text{SrP}_4\text{O}_{12}^{-2}$	4.7	0.2
$\text{Sr}^{+2} + \text{P}_2\text{O}_7^{-4} = \text{SrP}_2\text{O}_7^{-2}$	5.15	0.3
$\text{Sr}^{+2} + \text{P}_3\text{O}_{10}^{-5} = \text{SrP}_3\text{O}_{10}^{-3}$	6.7	0.5
$\text{Sr}^{+2} + \text{SO}_4^{-2} = \text{SrSO}_4$	2.49	0.10

$\text{Sr}^{+2} + 2\text{SO}_4^{-2} = \text{Sr}(\text{SO}_4)_2^{-2}$	2.86	0.2
$\text{Sr}^{+2} + \text{HCO}_3^- = \text{SrHCO}_3^+$	1.18	0.07
$\text{Sr}^{+2} + \text{CO}_3^{-2} = \text{SrCO}_3$	2.81	0.07
$\text{Sr}^{+2} + 2\text{CO}_3^{-2} = \text{Sr}(\text{CO}_3)_2^{-2}$	3.17	0.10
$\text{Sr}^{+2} + \text{C}_2\text{O}_4^{-2} = \text{SrC}_2\text{O}_4(\text{ao})$	2.73	0.15
$\text{Sr}^{+2} + 2\text{C}_2\text{O}_4^{-2} = \text{Sr}(\text{C}_2\text{O}_4)_2^{-2}$	3.4	0.2
$\text{Sr}^{+2} + \text{CH}_3\text{COO}^- = \text{SrCH}_3\text{COO}^+$	1.06	0.10
$\text{Sr}^{+2} + 2\text{CH}_3\text{COO}^- = \text{Sr}(\text{CH}_3\text{COO})_2$	1.63	0.10

## CESIUM

$\text{Cs}^+ + \text{Cl}^- = \text{CsCl}$	-0.31	0.13
$\text{Cs}^+ + \text{Br}^- = \text{CsBr}$	-0.18	0.2
$\text{Cs}^+ + \text{I}^- = \text{CsI}$	-0.50	0.3
$\text{Cs}^+ + \text{ClO}_3^- = \text{CsClO}_3$	-0.056	0.1
$\text{Cs}^+ + \text{ClO}_4^- = \text{CsClO}_4$	0.226	0.01
$\text{Cs}^+ + \text{BrO}_3^- = \text{CsBrO}_3$	0.009	0.024
$\text{Cs}^+ + \text{IO}_3^- = \text{CsIO}_3$	-0.118	0.024
$\text{Cs}^+ + \text{NO}_3^- = \text{CsNO}_3$	0.023	0.013
$\text{Cs}^+ + \text{SO}_4^{-2} = \text{CsSO}_4^-$	0.71	0.3
$\text{Cs}^+ + \text{P}_2\text{O}_7^{-4} = \text{CsP}_2\text{O}_7^{-3}$	2.3	0.1
$\text{Cs}^+ + \text{P}_3\text{O}_{10}^{-5} = \text{CsP}_3\text{O}_{10}^{-4}$	2.8	0.1
$\text{Cs}^+ + [\text{Fe}(\text{CN})_6]^{-3} = \text{Cs}[\text{Fe}(\text{CN})_6]^{-2}$	1.14	0.3

## 4. Speciation of radionuclides in seawater

The processes of hydrolysis and complexation by chloride, sulphate, bicarbonate, carbonate, and dissolved organic material determine the speciation of dissolved radionuclides in any oxidation state. Among organic matters the humic and fulvic acids are important [Choppin and Allard, 1985]. The examination of thermodynamic and environmental data indicates that speciation of Pu in marine waters is primarily governed by kinetic factors, and that interactions with organic compounds, such as humates, are important [Morse and Choppin, 1986].

The distinguishing features of the system "DiaNIK" include commentaries accompanying the recommended thermodynamic values, and a completely automated procedure for the information preparation intended for the computing of the equilibrium

composition of multicomponent systems, including aqueous species, over a wide T, P-range.

The first, preliminary results concerning computer modelling of chemical forms of Sr and Cs in seawater, which has been obtained using "DiaNIK" databases (see table) by method of Gibbs energy minimization, show, that in the seawater the single ions  $\text{Sr}^{+2}$  (71% of total strontium concentration) and  $\text{Cs}^+$  (88%) are predominant. The complexes -  $\text{SrSO}_4$  - 18%;  $\text{SrCl}^+$  - 10%;  $\text{Sr}(\text{SO}_4)_2^{-2}$  - 1% and  $\text{CsSO}_4^-$  - 18%;  $\text{CsCl}$  - 4%.

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**TEMPORAL EVOLUTION OF ARTIFICIAL RADIOACTIVITY IN THE NORTH AND  
BALTIC SEA AFTER THE REDUCTION OF THE SELLAFIELD DISCHARGES AND  
THE CHERNOBYL ACCIDENT**

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**Abstract**

Discharges of radionuclides from the Western European nuclear reprocessing plants at Sellafield and La Hague are transported into the North Sea by the prevailing current system according to the solubility of the nuclides. The concentrations of artificial radionuclides in the North Sea and the Baltic Sea during recent years followed the decrease of the discharges. As far as transuranics are concerned, enhanced levels can only be detected in water masses entering the North Sea.

The fallout from the Chernobyl accident in April 1986 deposited Cs-134 and Cs-137 mainly in the northern part of the Baltic Sea. These high concentrations decreased due to mixing of Baltic sea water and generated higher concentrations in less contaminated areas. It can be expected that the levels of Cs-137 in the Baltic will remain for a long period at higher levels than observed prior to the Chernobyl accident. The contamination from this source caused only a transient increase in North Sea water.

Part of the radionuclides released are deposited on sediments. The sediments of the Baltic Sea show a significantly higher specific activity than the sediments of the North Sea. This is due both to the higher areal deposition and to the higher adsorption capacity of the muddy sediments in the Baltic Sea. The highest levels of Cs-137 are still detected in the surface top layers of muddy sediments.

The dissolved radionuclides in the North Sea are finally transported along the Norwegian coastal current to the Arctic Ocean.

## Introduction

The North Sea and the Baltic Sea are surrounded by a number of highly developed countries. The population living on the drainage area of the North Sea is about 100 million and that of the Baltic Sea drainage area approximately 70 million inhabitants. As a consequence of the complete usage of the nuclear fuel cycle in these countries, these shelf sea areas receive also radioactive discharges from nuclear facilities. The most relevant sources of artificial discharges into these regional seas were the discharges from the reprocessing plants for spent nuclear fuel at Sellafield (UK) located at the Irish Sea and, to a lesser extent, Cap de la Hague (France). Due to the prevailing water mass transport in the Irish and North Sea the contaminated water was transferred into the North Sea and subsequently along the Norwegian coastal current to the Arctic Ocean. A minor part of the contamination could be observed before 1986 in the Baltic Sea which is a semi-enclosed sea area with limited water exchange capacity to the world ocean.

Discharges of nuclear waste water at Sellafield peaked in 1975 and decreased almost continuously to 1985. A further significant reduction of the discharges occurred from 1985 to 1986 when new waste water treatment facilities were installed. The discharges are documented in the annual reports by MAFF or BNFL (MAFF, 1993; BNFL, 1991). The liquid effluents from the reprocessing plant at La Hague into the Channel were less, compared to the Sellafield discharges over the entire period. However, this plant reduced its discharges as well during the last years.

## Concentrations of Cs-137 and Sr-90 in the North Sea

The radionuclides Cs-137 and Sr-90 behave almost conservatively in seawater. Their adsorption affinity to suspended particulate matter is rather limited. Therefore, the concentrations of these nuclides observed at various positions in the North Sea reflect the relative discharges according to the transport time of the contaminated water from the release points to the position of observation (Kautsky, 1973; Livingston et al., 1982; Nies, 1990). The different nuclide spectrum in the effluents from La Hague and Sellafield provided an opportunity to distinguish the water masses in the North Sea contaminated by Sellafield or La Hague, respectively. The activity ratios Sr-90/Cs-137 in the La Hague effluents were significantly higher than those of the Sellafield discharges. The northwestern and western part of the North Sea received only water contaminated by Sellafield, the southern part and the inner German Bight showed typical La Hague contamination. Mixing of these two water masses occurred at the eastern North Sea along the Jutland coast and in the Skagerrak.

Strong gradients in Cs-137 concentration were detected in the southern central North Sea during previous years with low Cs-137 activity concentration in the coastal current along the continental coast. Table 1 gives the averages of Cs-137 concentrations in surface water observed on stations in the central west-

ern North Sea, in an area from the eastern Channel along the Belgium-Dutch coast, and the German Bight. It is obvious that the most significant decrease of the concentrations occurred in the western central North Sea which received most of the activity of soluble radionuclides released at Sellafield.

Table 1:

Averaged activity concentrations ( $\text{Bq}\cdot\text{m}^{-3}$ ) of Cs-137  $\pm$  1 std. deviation (n = number of observations) in three areas of the North Sea; the areas are indicated in Fig. 2.

Area 1: Western Central North Sea;

Area 2: Eastern English Channel and waters along the Belgium-Dutch coast;

Area 3: Inner German Bight.

Time	Area 1 (n)	Area 2 (n)	Area 3 (n)
08/1980	173 $\pm$ 49 (31)	12 $\pm$ 1 (15)	29 $\pm$ 11 (8)
11/1981	114 $\pm$ 41 (41)	22 $\pm$ 4 (19)	23 $\pm$ 5 (8)
08/1982	141 $\pm$ 42 (42)	21 $\pm$ 3 (17)	26 $\pm$ 2 (7)
11/1983	142 $\pm$ 21 (21)	19 $\pm$ 4 (16)	21 $\pm$ 3 (6)
06/1984	108 $\pm$ 28 (40)	13 $\pm$ 3 (21)	18 $\pm$ 4 (8)
05/1985	73 $\pm$ 20 (21)	14 $\pm$ 2 (19)	16 $\pm$ 1 (6)
04/1986		17 $\pm$ 5 (11)	17 $\pm$ 4 (6)
06/1986		24 $\pm$ 17 (2)	164 $\pm$ 30 (5)
01/1987		13 $\pm$ 4 (11)	28 $\pm$ 9 (6)
07/1987			21 $\pm$ 7 (10)
10/1987	42 $\pm$ 18 (18)	12 $\pm$ 6 (16)	14 $\pm$ 3 (6)
07/1988			16 $\pm$ 3 (9)
11/1988	29 $\pm$ 6 (13)	9 $\pm$ 3 (16)	11 $\pm$ 2 (6)
04/1989		8 $\pm$ 1 (12)	11 $\pm$ 3 (8)
11/1989	20 $\pm$ 6 (14)	10 $\pm$ 3 (14)	11 $\pm$ 2 (9)
03/1990		9 $\pm$ 2 (13)	11 $\pm$ 0.6 (8)
11/1990	13 $\pm$ 5 (9)	11 $\pm$ 2 (16)	9 $\pm$ 1 (10)
02/1991		9 $\pm$ 2 (13)	9 $\pm$ 1 (8)

The areas mainly contaminated by the La Hague releases revealed a much lower Cs-137 contamination. However, a slight further decrease of Cs-137 concentrations was observed in these two regions as from 1988.

The higher levels of the Chernobyl fallout in April 1986 were detected in the inner German Bight (Area 3) (DHI, 1986; Nies and Wedekind, 1988) and were ascertained by the significant peak given for the average values in June 1986 in Table 1. The impact from that fallout was less in western parts of the North Sea, as the concentrations of Cs-137 in Area 2 show (Mitchell and Steele, 1988).

Cs-134 and Cs-137 derived from Chernobyl was removed from the North Sea by means of the current system prevailing in the North Sea. The values detected in the German Bight after 1987 were

even lower than those of previous years. Thus, the levels of Cs-137 observed in almost the entire North Sea since 1990 have reached levels almost as low as those which were observed at the period of the global fallout from the atmospheric nuclear weapon tests in the sixties. For several years the highest levels of Cs-137 were observed at the position  $55^{\circ}$  N  $006^{\circ}$  E. In this spot, the Sellafield contamination is preserved for longer periods because of reduced mixing processes with adjacent areas. Fig. 1 displays the distribution of Cs-137 in North Sea surface water in November 1990. The Sr-90 distribution observed during the same cruise is given in Fig. 2.

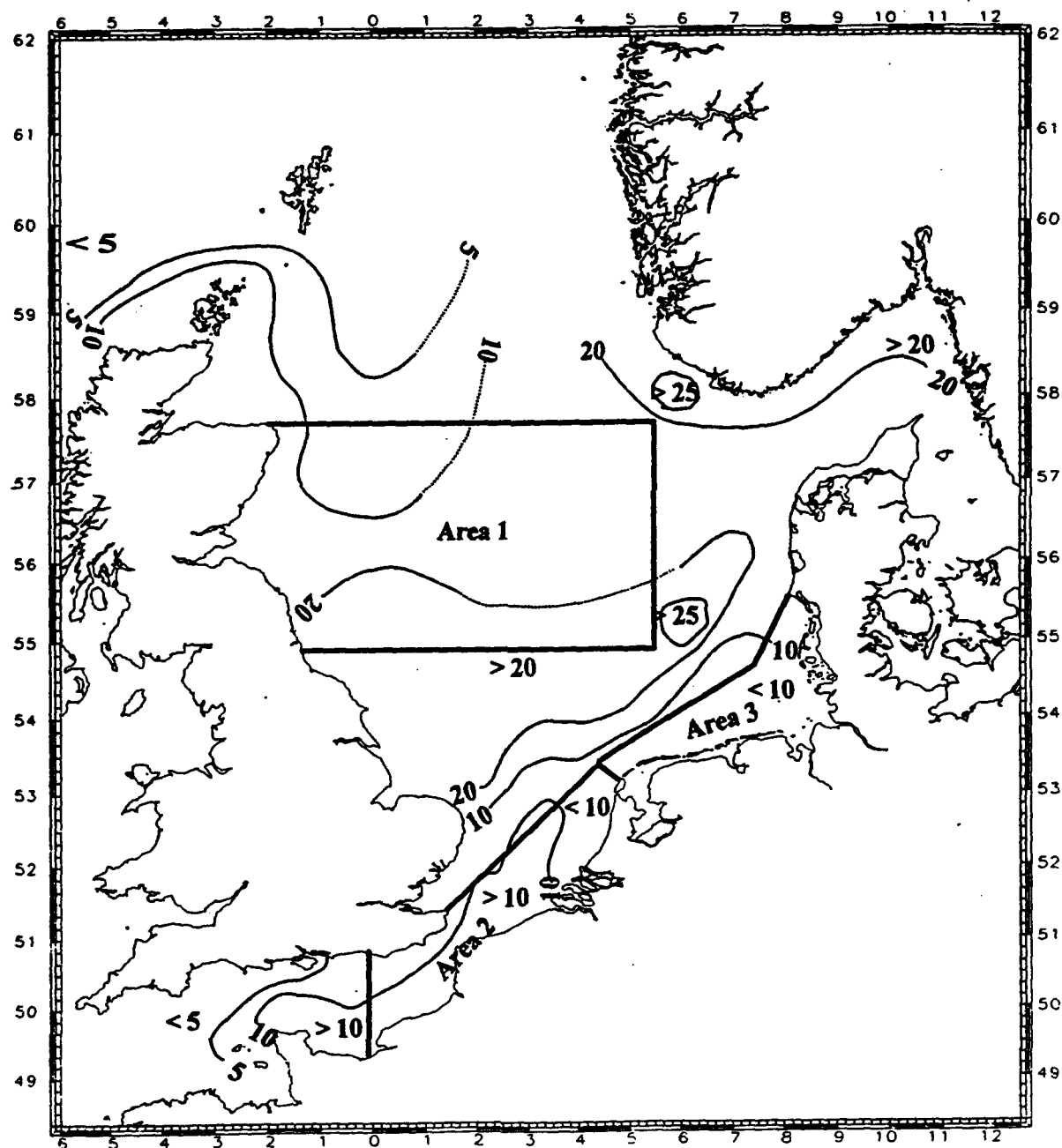
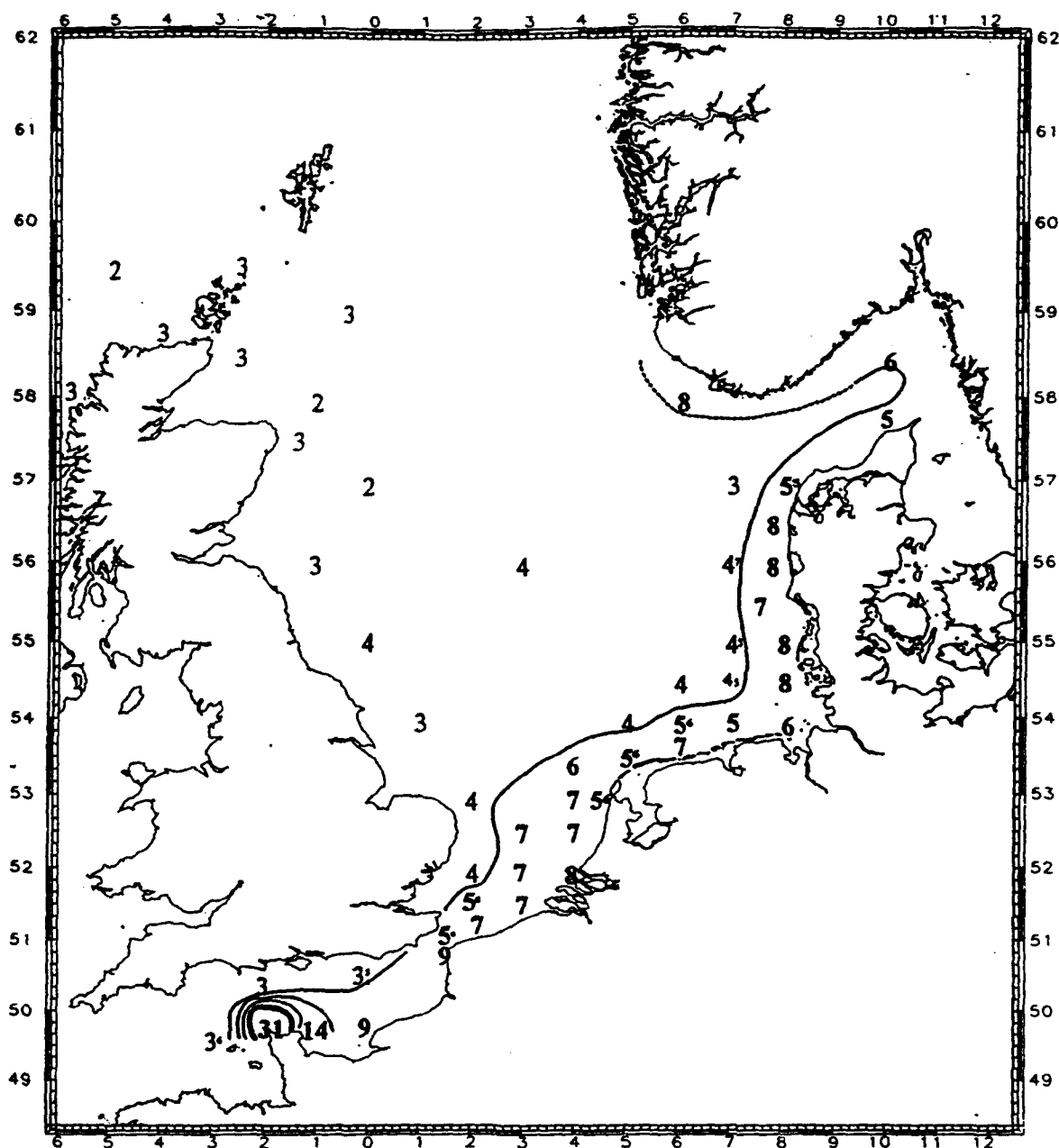


Fig. 1  
Distribution of Cs-137 ( $\text{Bq} \cdot \text{m}^{-3}$ ) in surface water of the North Sea in November 1990.

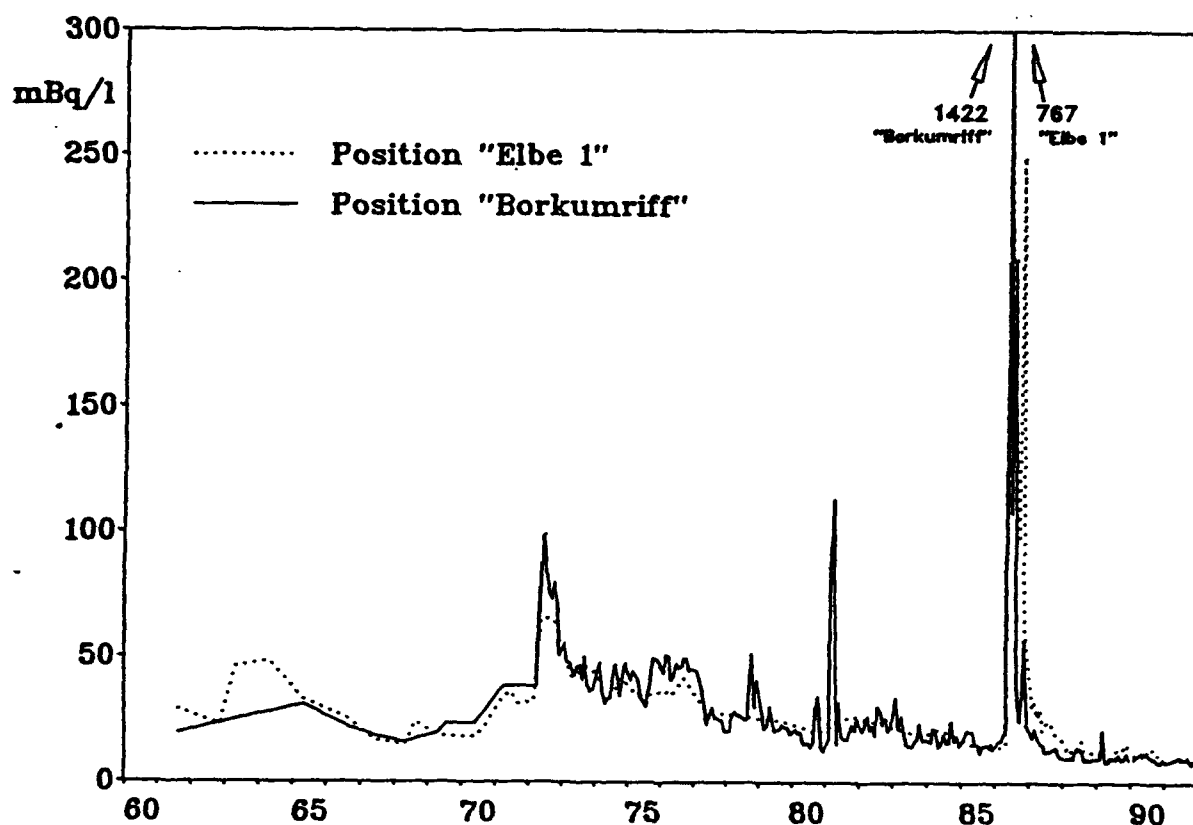


**Fig. 2**  
Distribution of Sr-90 ( $\text{Bq} \cdot \text{m}^{-3}$ ) in surface water of the North Sea in November 1990.

The activity concentration of Sr-90 in North Sea surface water in 1990 was enhanced in waters contaminated by La Hague effluents, while the concentration in the western and northern North Sea was only slightly above the values expected by global

fallout levels in the Northeast Atlantic surface waters. The discharge point at La Hague can easily be identified in Fig. 2 by the elevated levels which are reduced by dilution during the transport through the Channel into the North Sea.

From the beginning of the sixties, the temporal trend of the Cs-137 activity concentrations at two positions in the German Bight are given in Fig. 3. At these two positions, water samples for analysis were taken on a monthly basis from the seventies onwards.



**Fig. 3**

Temporal trend of the Cs-137 activity concentration ( $\text{Bq} \cdot \text{m}^{-3}$ ) at two positions in the German Bight from 1961

The peak in spring 1981 in Fig. 3 is a result of strong north-westerly gales which forced water from the central North Sea into the inner German Bight. This water contained a higher Cs-137 concentration due to Sellafield contamination.

The Chernobyl fallout resulted in an extreme peak in 1986. However, due to dilution in the water column the high values decreased to values measured before the accident within one year. The concentrations at station "Elbe 1" decreased less quickly, due both to limited water flushing time at this position and run-off from the river Elbe from areas significantly contaminated upstream in Chechoslovakia.

### Concentrations of Cs-137 in the Baltic Sea

Artificial radioactivity in the Baltic Sea originated from nuclear weapon tests, from inflow of North Sea water, and, since 1986, from the Chernobyl fallout. The typical level of Cs-137 in the Kattegat before 1986 was of 25 to 30 Bq·m<sup>-3</sup> in the surface water and 60 to 90 Bq·m<sup>-3</sup> in deeper layers with higher salinity. Sr 90 was determined from 18 to 20 and 20 to 23 Bq·m<sup>-3</sup>, respectively. The concentrations of Cs-137 declined with decreasing salinity to the northern part of the Baltic revealing the source being the North Sea. In the northernmost part of the Baltic Sea, Cs 137 concentrations were only from 6 to 10 Bq·m<sup>-3</sup> in surface water. Sr 90 was more homogeneously distributed in the water column of the Baltic with values of about 20 Bq·m<sup>-3</sup>. In the Bay of Bothnia these values decreased to 10 Bq·m<sup>-3</sup> in the surface water and about 16 Bq·m<sup>-3</sup> in deeper layers beneath the halocline (IAEA, 1986; Nies, 1988).

The Chernobyl fallout in April 1986 changed the inventory of Cs-137 in the Baltic Sea significantly. This fallout can be identified by its characteristic Cs-134/Cs-137 activity ratio. It turned out that the Baltic Sea was affected very unevenly. The main fallout occurred in the Sea of Bothnia by contaminated precipitation. Its distribution and change in subsequent years was scrutinized in international cooperation by an expert group of the Baltic Sea Environment Protection Commission - Helsinki-Commission (HELCOM, 1989; Ribbe et al., 1991).

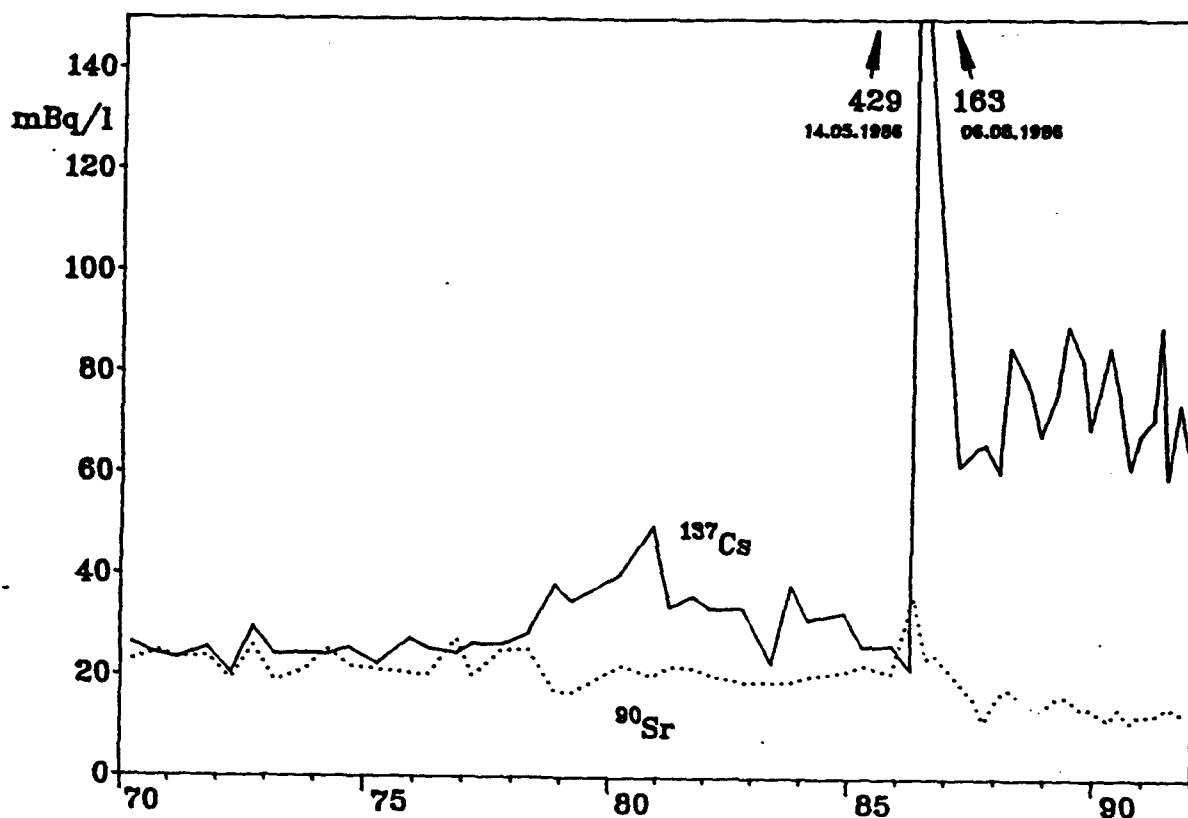
A time series record of Cs-137 and Sr-90 at one position located in the western Baltic Sea from 1970 to 1991 is given in Fig. 4. This position is very sensitive for the exchange of water masses between the Baltic and the North Sea via the Skagerrak and Kattegat area. The peak values of Cs-137 in 1980 are a consequence of the peak discharges of this radionuclide at Sellafield in 1975. Thus, the transport of Sellafield derived water takes about five years to pass from the Irish Sea through the North Sea to the entrance of the Baltic Sea. The decrease in subsequent years is a consequence of decreasing discharges at Sellafield.

The deposition of the Chernobyl fallout is clearly pronounced by the peak in May 1986. However, due to dilution of the surface water contamination in the water column the values dropped until April 1987 when the lowest activity concentration was detected. From 1987 onwards, there exists a seasonal variation of the Cs-137 concentration with increasing values during spring and decreasing values during late summer/fall with a slight tendency of a general increase until 1990.

The seasonal variations reflect the inflow and outflow of water through the Danish Straits. In spring, when snow melting and higher rain deposition in the drainage area of the Baltic Sea occur, the surface water moves into western direction. This water carries the higher contamination of the fallout from the northern Baltic area. During summer and fall the water moves in eastern direction with less contamination from the Chernobyl



fallout in Kattegat water. The slight general increase up to the year 1990 seems to be an effect of water transport from the northern Baltic to the less contaminated water via the Baltic Proper towards the Belt Sea. It can be expected that the enhanced level of contamination in the Baltic will last for a longer period corresponding to the water exchange of the Baltic Sea with the North Sea of about 25 to 30 years.

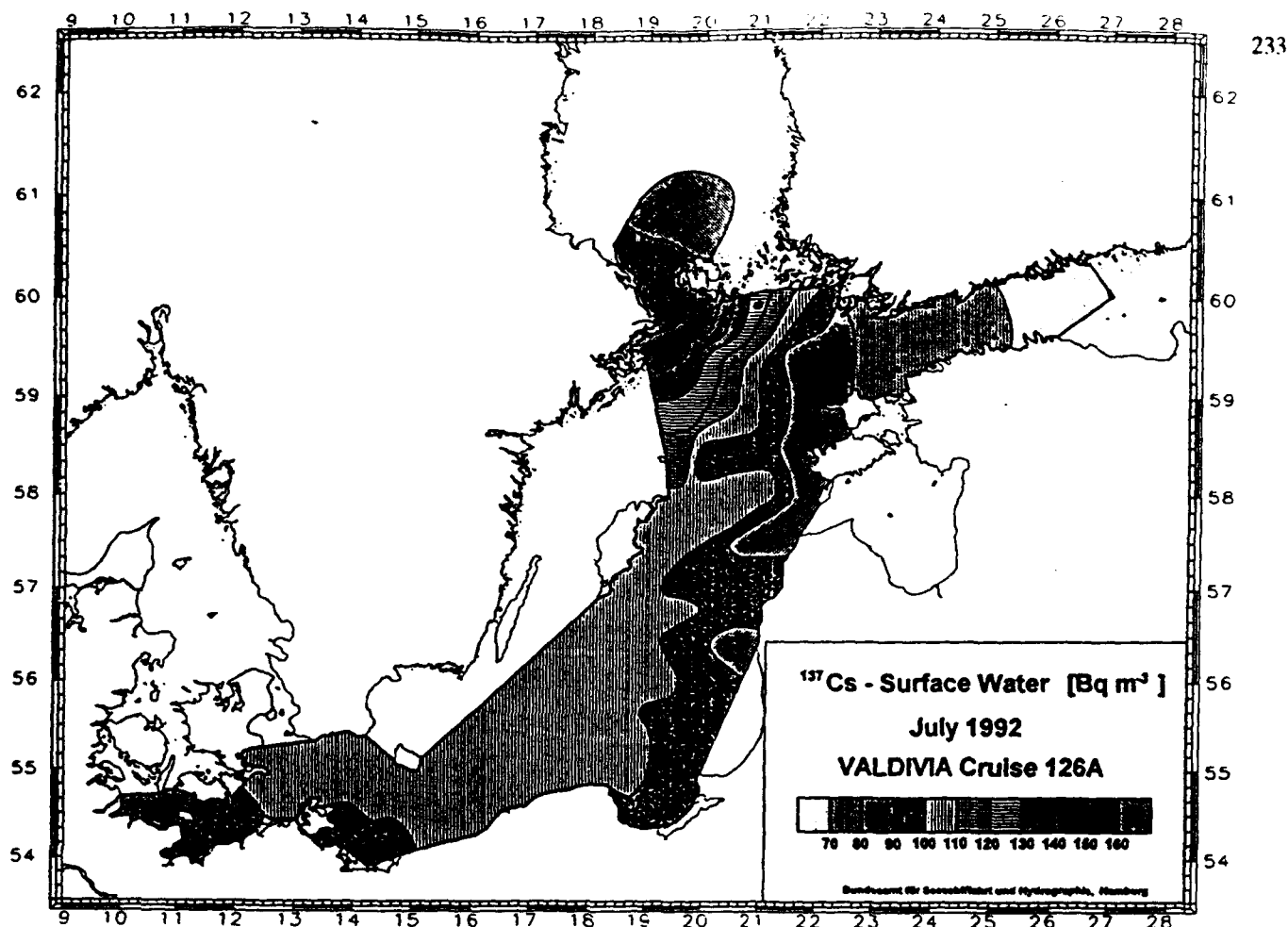


**Fig. 4**

Temporal trend from 1970 to 1991 of the Cs-137 and Sr-90 activity concentration in surface water at "Schleimündung", a position located in the western Baltic (54° 40'N 010° 05'E).

The outflow of contaminated water from the Baltic Sea into the North Sea causes elevated Cs-137 levels along the Norwegian coast (Fig. 3).

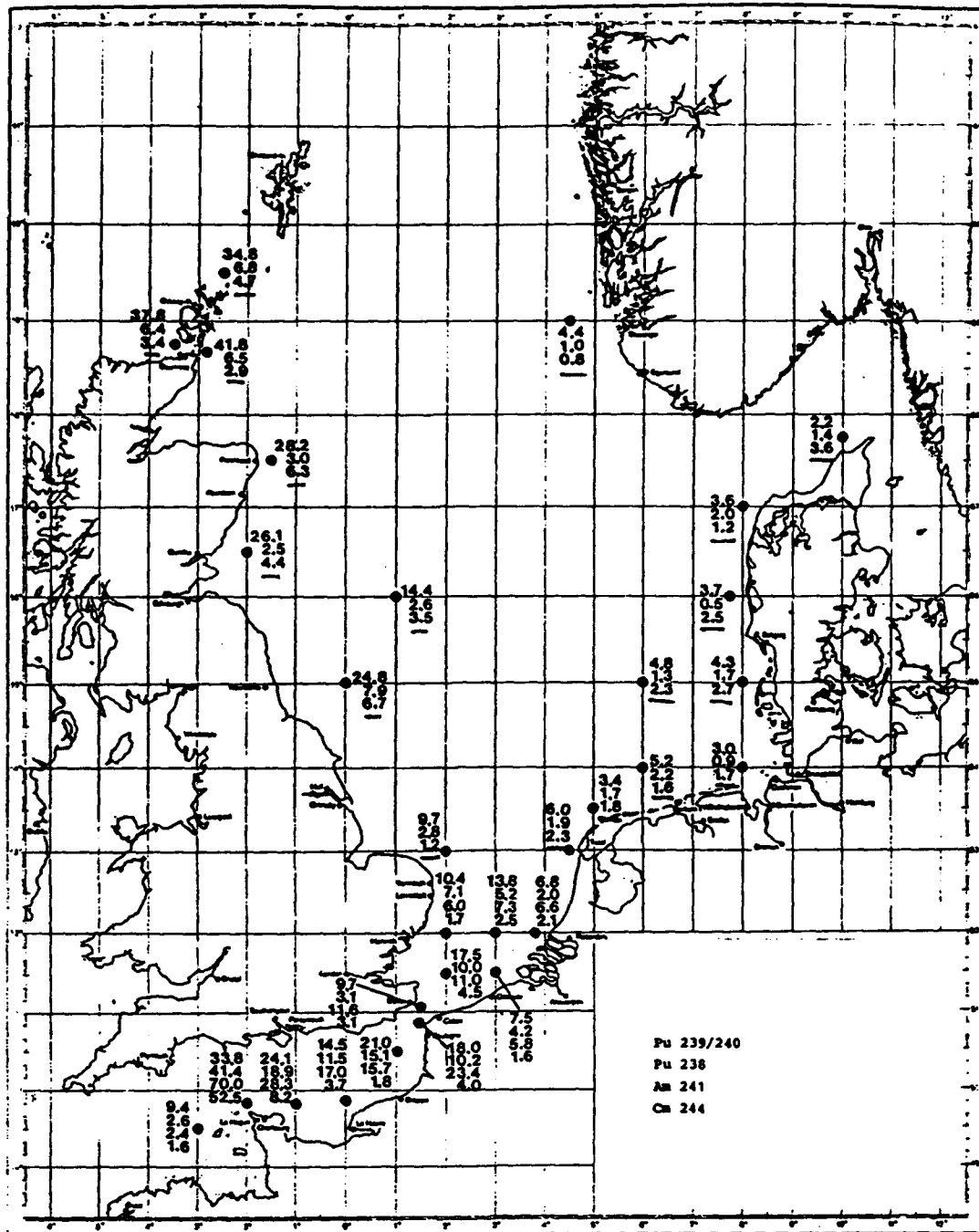
Recent cruises to the Baltic Sea show more homogenized surface contamination in the Baltic Proper. Fig. 5 displays the survey of Cs-137 in surface water during summer 1992. The Chernobyl derived fallout has reached deeper layers beneath the halocline. However, the most pronounced levels can still be measured in surface layers.



**Fig. 5**  
Distribution of Cs-137 ( $\text{Bq}\cdot\text{m}^{-3}$ ) in surface water of the Baltic Sea in July 1992

## Concentrations of Transuranics in the North Sea

We studied transuranic nuclides such as Pu-239/240, Pu-238, Am-241, and Cm-244 in North Sea water during several cruises. It is well known that transuranic nuclides are behaving more particle reactive than Cs-137. Therefore, it could be expected that transuranic nuclides would be removed from the water column during their migration with the water masses through the North Sea. The highest concentration of all transuranics were detected at the discharge area at La Hague for several years and the highest Pu-239/240 concentrations were measured in the Pentland Firth (north coast of Scotland). The latter are explained by remaining contamination on Pu-isotopes from Sellafield discharges. It is not assumed that these elevated levels are related to discharges from the nuclear facility at Dounreay, Scotland, because the same trend was observed west of the Dounreay area. Cm-244 could not be detected in this region as well as in other regions of the central North Sea.



**Fig. 6**

Transuranic nuclides Pu-239/240, Pu-238, Am-241 and Cm-244 (mBq·m<sup>-3</sup>), respectively, in surface water of the North Sea in November 1989.

The Am-241 concentrations were by far lower than the Pu-239/240 concentrations. This indicates the higher adsorption affinity to particles for Am-isotopes relatively to Pu-isotopes. It is very likely that Am-241 was removed from the water column during the transport from the Irish Sea to this region. Fig. 6 gives one example for the distribution of transuranics over the North Sea in November 1989.

In general, the concentrations decline with distance from these positions. The source of the transuranics in the Channel is unequivocally recognizable on the basis of the activity ratios which are indicative for releases from the nuclear fuel cycle. The concentrations of Pu-238 and Am-241 are almost similar or even higher than the Pu-239/240 values. However, during the passage through the Channel and, subsequently through the North Sea, the water becomes depleted primarily from Am-241 and Cm-244. In the German Bight and finally the Skagerrak, the concentrations of Pu-239/240 and Am-241 are similar to those measured in Atlantic surface waters. However, the Pu-238 activity concentrations indicate that the main origin of the Pu-isotopes is the release from this reprocessing plant. The activity ratios Pu-238/Pu-239 are still considerably above the ratio of the global fallout of about 0.04.

### Conclusion

During the last years, a significant temporal trend and spacial pattern of the contamination of the North Sea of artificial radionuclides was observed. Sellafield derived Cs-137 dominated the main inventory in the North Sea. According to the decrease of the discharges from 1975 onwards the contamination of Cs-137 followed the decline of the discharge in some time delay according to the water mass transport time into the North Sea. During recent years the concentrations of artificial radionuclides have reached values as low as before the seventies, when the main source of Cs-137 and Sr-90 was the global weapon fallout. Prior to the Chernobyl fallout, Cs-137 was transferred towards the Baltic from the North Sea with the consequence that the Cs-137 concentration correlated to the salinity.

The Chernobyl fallout into the North Sea was removed from this sea area within a period of about two years, while the limited water exchange capability of the Baltic Sea with the world ocean will prevent a rapid decrease of the Cs-137 concentration in the significantly contaminated water.

The contamination of North Sea water by transuranics can unequivocally identified by the activity ratios, which are indicative for releases from reprocessing plants.

The contamination of artificial radionuclides in the North Sea will be transported to Arctic regions along the Norwegian coastal current. This contamination source and its temporal change has to be taken into consideration for investigations to be carried out in parts of the Arctic Ocean.

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Organization and Technical Facilities of the Radiation  
Monitoring of Sunken Objects with Radioactive Components

In 1992 CDB ME "Rubin" and associated Russian research organizations put forward an initiative to establish the marine radiation and environmental monitoring system which is intended to monitor radiation hazardous objects on sea and ocean beds. The conclusion on the advisability and necessity of establishing such a system was made proceeding from the results of works in 1990 to 1992 connected with the nuclear submarine "Komsomolets" which sunk in 1989, and primarily from the results of marine expeditions carried out in the Norwegian Sea.

The complex investigations of the radiation and environmental situation in the vicinity of radiation hazardous objects lying on the sea and ocean bed due to various reasons and of the technical conditions of these objects similar to the carried out expeditions to the nuclear submarine "Komsomolets" in 1991 and 1992 and the expedition planned to be carried out in July-August 1993, are absolutely necessary but still unique experiments which require significant amount of efforts, funds and time. In order to optimize the expenses and to increase scientific and technical level of such works the establishment of the marine radiation monitoring system is extremely important.

The accomplished radiation and environmental researches connected with the SSN "Komsomolets" laid down the foundation of the systematic radiation monitoring in the site of the submarine, and they can be considered as the initial stage of the establishment and one of the possible forms of realization of the marine radiation

monitoring for that and similar objects.

The scientific devices and equipment developed and used for the examination of the radiation situation on the SSN "Komsomolets" and in the surrounding medium could become the basic elements of technical facilities of the marine radiation monitoring system.

The prerequisites for initiating works on the development of the marine deep water radiation and environmental monitoring system are as follows:

- the experience gained during 1990 to 1992 in works related to the submarine "Komsomolets";

- the existing system of the extensive oceanographic support;

- proven under actual conditions new highly sensitive deep water radiation measuring devices and other required facilities (sorbents, samplers);

- availability of proven methodics for analysis the obtained measurement and sampling data;

- availability of certain scientific experience in evaluating radiation effects on the environment and in predicting their trends in the marine environment proceeding from the preliminary initial data and those obtained during actual measurements;

- availability of a well organized and successfully working team of scientists and designers of various fields of activities.

The concept of the marine radiation and monitoring is developed proceeding from two main assumptions. First, there is a substantial number of radiation sources on the sea-bed of the World ocean, and their number will obviously increase. Second, politicians, scientists and the general public would like to know what is the nature of this radiation danger and how great it is. A well proved answer to this question, convincing both for governmental organizations and for independent experts, is decisive for the fate

of a sunken object, e.g. a nuclear submarine, or the fate of 239 critical groups of population in certain regions. Finally, radiation sources on land are strictly registered and continuously monitored with the help of rather complicated and broad monitoring systems, while radiation sources on the sea-bed are practically out of reach of radiation monitoring.

A number of major general proposals of the International Commission on Radiological Protection (Publications Nos 36, 37, 43, 60 and other) is taken as the basis for defining the aims, tasks and methods of the marine radiation monitoring system establishment. For this case, these definitions are brought to the following:

- the most important problem arising due to a radiation release accident (incident) or in a case of its prediction with more or less considerable level of probability, is the selection of a strategy of intervention to eliminate the consequences of the accident or to prevent the accident, therefore, before the programme of intervention is initiated it should be demonstrated that the proposed intervention will be justified, i.e. do more good than harm and that the form, scope and duration of the intervention have been chosen so as to optimize the protection. The difference between the intervention, e.g. the recovery of a sunken object, and non-intervention when expressed in the same terms, e.g. costs, including social costs with an allowance for fear and anxiety, should be positive and should be maximised by settling the details of the intervention strategy;

- quantitatively harmful effects of possible exposure to radiation are measured by "detriment". The detriment is defined as the mathematic expectation of harm at an exposed group of people with allowance for probable various deleterious effects and the measure of severity of those effects as well as anxiety and worries



of people with regard to probable radiation risk and any deleterious consequences for their comfort due to restrictions resulting from exposure to radiation;

- the monitoring is defined as measurement of radiation or concentration of radionuclides to estimate (or monitor) the effect of external radiation or of a radioactive substance including measurement interpretation based on model describing quantitative relations between measured and designed values. It is emphasised that both an analytical model and actual measurements are equally important in the monitoring programme (the model to interpret the results of measurements permits to select appropriate methods of measurements).

Proceeding from the stated above, the aim of the marine radiation monitoring system is defined as the forecast for radiation detriment and the information support for decision making on actions regarding radiation hazardous marine objects.

The achievement of this aim should be provided by the solution of the following problems:

- carrying out of radiation monitoring, permanent estimation of the radiation situation;

- determination, prediction and monitoring of technical conditions of the object;

- estimation of the dynamics of the radiation situation and technical conditions of the object and the adequacy of the prediction models;

- carrying out of periodic corrections of the prediction estimations proceeding from the results of natural radioactivity measurements and the examination of technical conditions of the object;

- providing for the radiological protection of the personnel

during the inspection of the object and protection works;

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- formation of the environmental data base, creation of the data bank on the ecology of man and environment;

- information output to support the control decision making on the protection works (procedures).

The broad spectrum of probable radiation hazardous events in sea with objects containing radioactive components is stipulated by the necessity to establish the multipurpose marine radiation system as it should cover such events as sinking of a nuclear powered ships (with reactor in operation or stopped, with hull and nuclear power plant destroyed or without any damage to the hull and the nuclear power plant), fall of a nuclear powered space object, sinking of a ship for liquid and solid radioactive waste treatment and transportation, partial or complete loss of sealing in a sunken container with radioactive wastes, non-authorized disposal of radioactive wastes, burial of nuclear ammunition at sea (with a carrier - a ship, an aircraft, a missile - or without it, destroyed or not destroyed) etc. With any of these events it is possible that single-time or continuous release of radioactive substances to the sea will take place.

Obviously, the International Register of radiation sources lying on the sea-bed of the World ocean, will be developed. The marine radiation monitoring system, which is planned to be established in Russia, can be considered to be a required integral part of the Register which provides continuous supply of required information for corresponding objects and sea areas based on the common, internationally recognised, methodology.

Figure 1 shows the principal functional diagram of the universal marine radiation monitoring system.

The structure of each subsystem shown on this diagram has its

own degree of a scientific novelty and a scope of non-solved problems. For example, during the development of the subsystem "Ecology of Man" it is supposed to implement the main definitions of the up-to-date approaches of the International Commission on Radiological Protection and the United Nations Scientific Committee on the effects of atomic radiation related to normalization of radiation factors and estimation of the degree of radiation danger for situations when radiation sources lie on the seabed of the World ocean. It will be necessary to develop a package of programmes required to forecast deleterious consequences of exposure for population due to a single-time or permanent contamination of sea products taking into account direct and indirect foodstuff chain - consumption of contaminated fish, the use of desalinated water, the use of sea products in agriculture, e.g. to produce fertilizers etc. While developing the subsystem "Ecology of Environment" the intention is to develop criteria of radiological capacity of the marine media which will take into account the exposure of not only man, i.e. hygiene aspects, but the problem of the hydrobionts protection also. In other words, ecological norms are supposed to be developed which should reflect the degree of protection for biological species including those which do not participate in formation of foodstuff chains but which are needed to preserve ecological equilibrium in a region. It should be taken into account in this context, that since sea products are used in agriculture (feeds, fertilizers), it is necessary to include not only hydrobiocoenoses but also biogeocoenoses.

For the marine radiation monitoring system, the obtaining of the required scope of high-quality information which can provide for accomplishing the tasks and aims of the system both technically and organizationally, should be adapted for the following probable

situations:

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- express-analysis and estimation of the radiation situation in the area of a marine nuclear incident after several hours or days;

- carrying out of casual full-scale measurements in the site of a single-time release of radioactive substances from the object into the sea;

- carrying out of long-duration and continuous or systematic measurements during permanent release of radioactive substances from the object into the sea;

- carrying out of measurements from the point of view of ecology of environment, ecology of man and radiological protection of personnel during protection or research works under stable radiation;

- carrying out of measurements under conditions of deterioration of the radiation situation during protection operations.

A possible variant of organization of the marine radiation monitoring system which can meet the mentioned situations is shown in figure 2.

It is evident that technical facilities providing for obtaining of the required information can not become a closed-circuit system, but should be an open, i.e. adaptable to flexible modifications, combination of portable and stationary measuring devices. Stationary devices form a subsystem of automated radiation monitoring which works continuously and in independent mode. Portable devices are used periodically (for inspections) or episodically (during protective operations).

The suggested marine radiation monitoring system allows to solve the problem of providing data on radiation situation both in case of the incident with the object containing the nuclear

components and the necessity to carryout long-duration or systematic observations at sites of known positions of radiation hazardous objects.

In case radionuclides, gamma-emitters, are released to the environment, it is possible at present to organize continuous long-term monitoring. It seems to be interesting to use for these purposes devices of the system of sorbtion radionuclide concentration out of sea water with a spectrometer detector for continuous measurements of gamma emission mounted into it. It should be noted that accurate quantitative information with such devices can be obtained actually only for caesium isotopes. As to the monitoring of release from the object of radionuclides, the detection of which is possible only after radiochemical analysis in laboratories, it is necessary to periodically take samples of water, soil and hydrobiontes, to install and remove sorption units, and all these should be organized at sites of nuclear warheads on the sea-bed, of sunken nuclear submarines, sunken space objects with plutonium thermal-electric generators and other similar objects.

Within the frames of the marine radiation monitoring system it is necessary to provide for synchronised obtaining of information on radiation factor effects as well as on other factors required for forecasting estimations which include oceanographic parameters as well as data on hydrometeorological and geomorphological conditions at the area of the object.

It seems reasonable to develop three variants of measuring devices of the marine radiation monitoring system adapted, respectively, to operative investigations (express-analysis), single-time full-scale detailed investigations, long-term continuous or systematic monitoring of radiation effect on the environment in the water area of radiation hazardous objects lying on the sea-bed. All

systems may have some common technical facilities but each will be equipped with specific technical facilities specified, first of all, by the problem being solved.

The functional diagram of the interaction of elements of the marine radiation monitoring system is shown in figure 3.

In order to be able to accomplish operative investigations of the situation at the place of a nuclear incident, high-speed facilities for shipment of portable instrumentation to the accident area are required, for example, aircraft, equipped with adequate radiation-physical instrumentation systems for remote measurements of radiation fields over the water area and with air-deployed probes with radiation measuring instruments, immersion dosimeters, in particular.

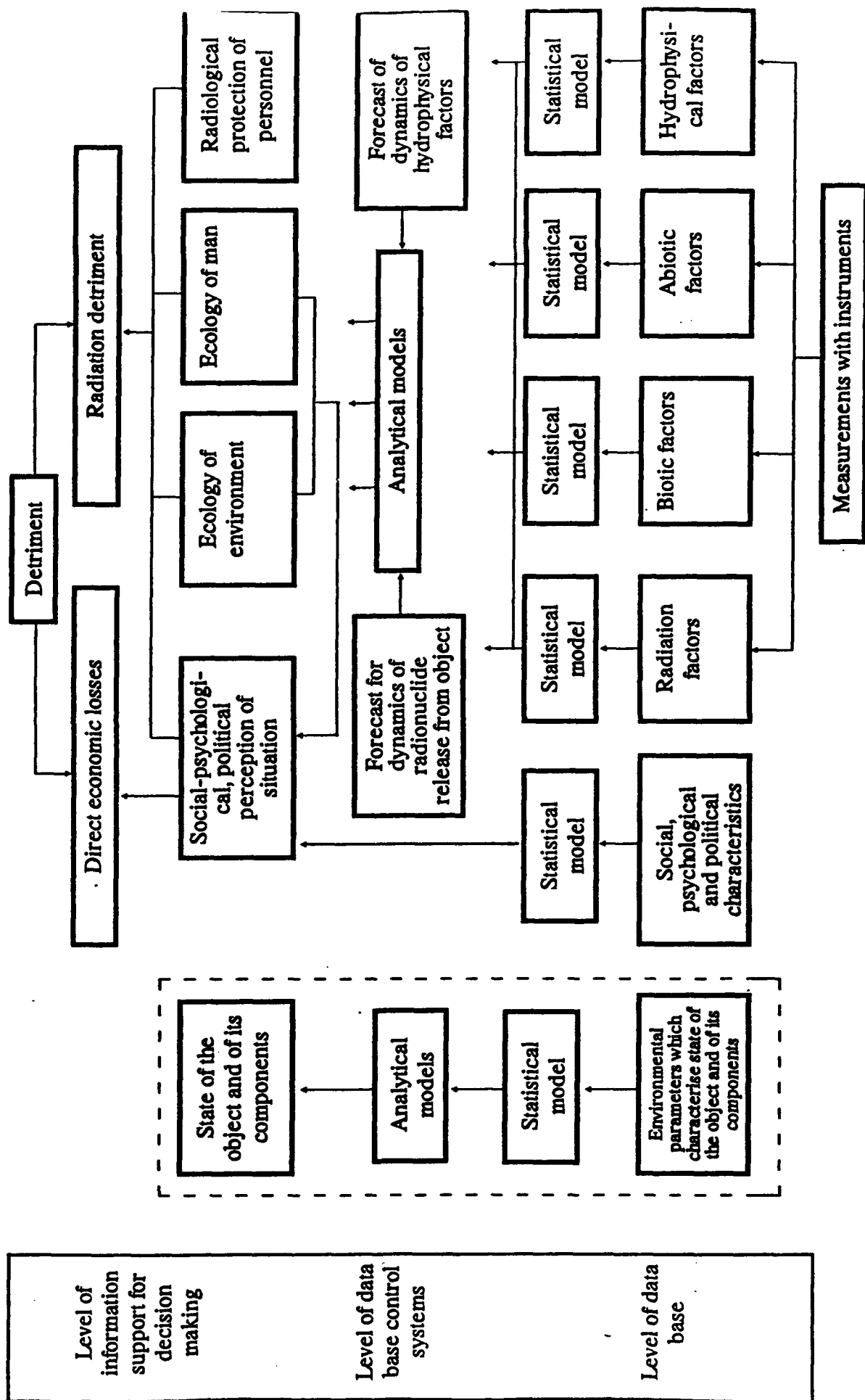
For carrying out of full-scale detailed investigations, which as a rule are casual owing to the complexity of their organization and high cost, it is necessary to use deep-water vehicles as a facility to deliver instrumentation and sampling units to the inspected object and means for inspection of technical conditions of the object. A significant number of research methods and devices in this case can be orientated to the use of deep-water vehicles from specially equipped ships including, in situations when it is permissible from the radiation safety criteria point of view, deep-water manned vehicles. It should be noted that the use of deep-water manned vehicles "Mir" in investigations of the sunken nuclear submarine "Komsomolets" in 1989 to 1992 had shown their high efficiency and big opportunities in carrying out such works.

It is reasonable to carry out long-term continuous or systematic observations using stationary self-contained bottom stations equipped with adequate instrumentation and sampling units, means of information acquisition and transmission through

communication channels as well as with means of sample transfer for the subsequent laboratory analysis. It is desirable to carry out the installation of such stations and reception of samples and measuring data practically from any not specially equipped ship.

The determination of the actual composition and design features of the mentioned complexes of technical facilities will be a very important stage of the planned works on establishment of the marine radiation monitoring system. At present Russia has practically all components required for creation of complexes of facilities of the marine radiation monitoring systems, but realization of such complexes as well as of the marine radiation monitoring system as a whole, taking into account very strict requirements which they should meet, could be considerably accelerated with the international cooperation in establishment and subsequent operation of the system. In case such a system takes the shape of an international structure, then such an organizational-and-technical structure could, obviously, function under the patronage of such authoritative international organization as the International Commission of Radiological Protection, International Agency on Atomic Energy etc., and should provide them as well as governmental and public structures with the unbiased information about the radioactivity in the World ocean they need so much.

# PRINCIPAL FUNCTIONAL DIAGRAM OF THE MARINE RADIATION MONITORING SYSTEM





# ORGANIZATION OF MARINE RADIATION MONITORING SYSTEM

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Elements of monitoring structure, methods of obtaining information		Ways to accomplish the tasks of monitoring		
		Active inspection of accident area		Continuous and periodical monitoring of known zones and radiation sources
		Express-estimation of the situation	Full-scale inspection of the accident area	
Carriers of technical facilities for obtaining information and its processing	Aircraft of environmental reconnaissance	Main mission	Possible involvement	Possible involvement
	Ships of environmental inspection		Main mission	Main mission
	Shore bases for preparation of ships, aircraft, underwater vehicles, self-contained stations and devices	Provides all mission		
	Base laboratories for radiometric and spectrometric measurements and for physical-and-chemical and radiochemical analysis	Provides all mission		
	Information and analytical centres for information processing, situation forecast, making decisions on situations in the regions of the World ocean	Provides all mission		
Technical facilities (devices) for obtaining information	Manned underwater vehicles	Possible involvement	Main mission	Possible involvement
	Unmanned underwater vehicles	Possible involvement	Main mission	Main mission
	Deployed (immersion) stations for long-term operation			Main mission
	Devices for single-time sampling (of sediments)	Possible involvement	Main mission	Main mission
	Radiation-physical hardware systems	Main mission	Main mission	Main mission
	Physical and chemical hardware systems		Main mission	Main mission
Methods of obtaining information	Remote measurements of radiation-physical (and aerosole-gas) fields over the water area	Main mission	Main mission	Possible involvement
	Remote measurements of physical (hydrological, hydromechanical) fields in marine environment		Main mission	Main mission
	Contact measurements of fields of substances and products injected at accidents in World ocean		Main mission	Main mission
	Sampling of ocean media		Main mission	Main mission
	Sampling of ocean media while exposing sorbents to accumulate specific substances		Main mission	Main mission
Dynamics of obtaining information	Receiving (output) of information at the time of measurement	Main mission	Main mission	Main mission
	Receiving (output) of information after processing of results of measurements and sample analysis aboard carriers of technical facilities		Main mission	Main mission
	Receiving (output) of information after processing of results of measurements and sample analysis at base laboratories		Main mission	Main mission
	Output of information by information and analytical centres	Main mission	Main mission	Main mission

# FUNCTIONAL INTERACTION OF ELEMENTS OF THE WORLD-OCEAN MONITORING SYSTEM

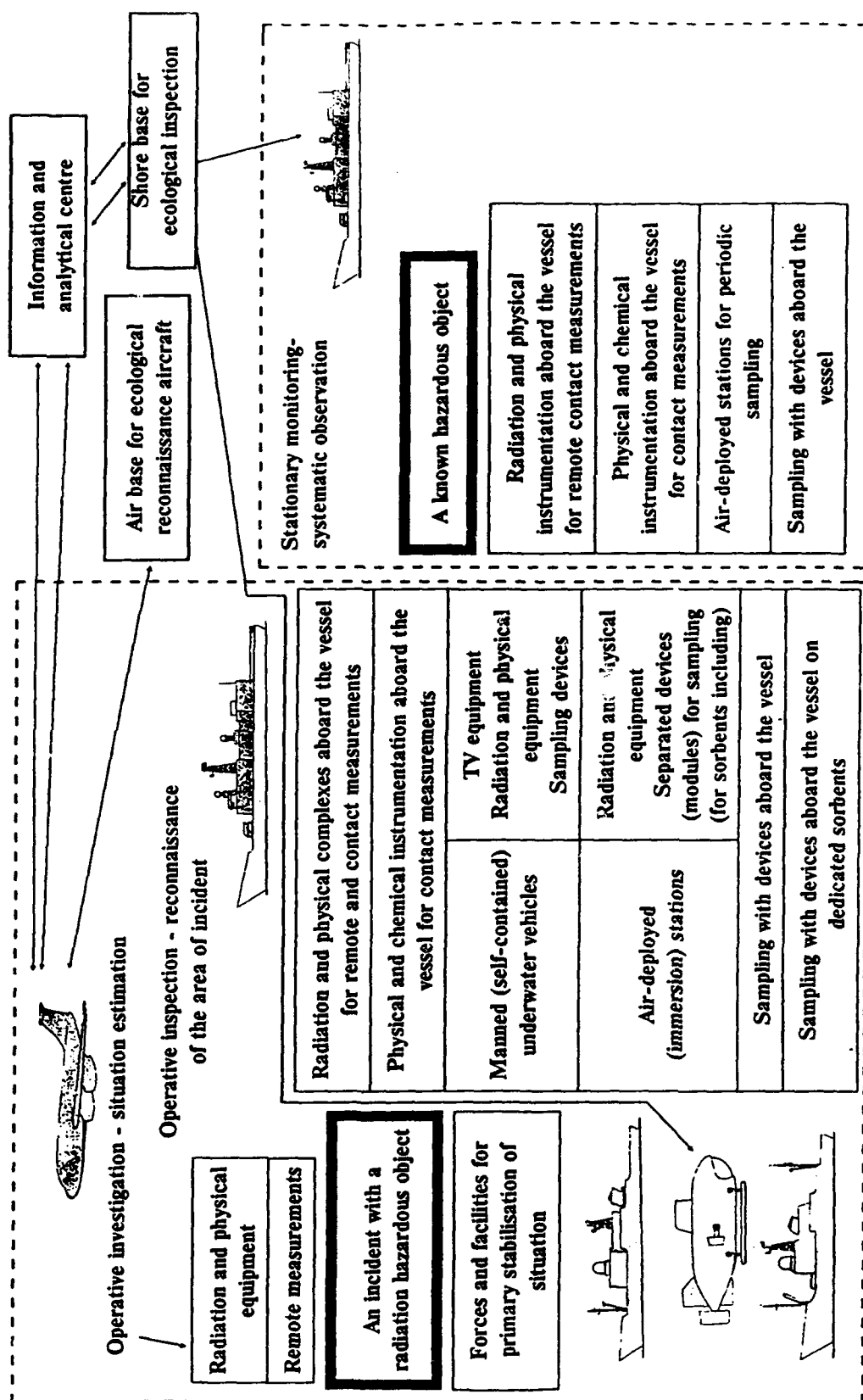


Fig. 3



**FORMATION FEATURES OF THE STRUCTURE AND VARIABILITY OF THE  
HYDROLOGICAL PROCESSES OF THE KARA SEA AREA**

The importance of the studies of the hydrometeorological processes in the Kara Sea has been lately related with the problem of radioactive waste dumping to its bottom by the former Soviet Union and its possible penetration into the marine environment. The international conference "Actual and potential consequences of dumping of radioactive waste", held in Oslo (Norway) in February of 1993, stressed a great importance of the studies of modelling and forecasting of the hydrophysical processes and phenomena, which can account for the transfer, transformation and accumulation of the radioactive substances from the possible pollution sources.

The Arctic and Antarctic Research Institute (St. Petersburg, Russia) has been conducting regular hydrometeorological studies of the Kara Sea and of the river mouth areas for more than 50 years. By the present time a vast amount of data on the structure and variability of the hydrophysical fields has been accumulated, some features of which, important from our viewpoint in relation with the radioactive contamination problem will be considered below.

The Kara Sea is one of the Siberian Arctic Seas. It has an extensive water border with the Arctic Ocean, its largest part situated at the land bank, having the characteristics of the marginal land seas [1]. Its area [2] is equal to 883 000 sq. km, its volume is 98 000 cu. km, with 81.8 % of the area and 77.5 % of the volume constituting the shelf zone with depths up to 200 m. Mean depth of 111 m, maximum depth of 620 m are observed in its northern part at the point with the coordinates 80 26' N and 71 18' E.

One of the characteristic features of the natural conditions of the Kara Sea appears to be a complex bottom topography. In the southern and eastern parts of the sea the depths up to 100 m prevail. Here the sea bottom is crossed by numerous small deepenings, divided by the ridges of a different height. A relatively level bottom is found in the central regions. The largest depths are observed in the deep-sea troughs of St. Anna and Voronin in the western and north-western parts of the sea, between which there is a Central Kara rise with the depths less than 50 m.

The high-latitudinal location of the Kara Sea, direct connection with the Arctic Ocean and the Novaya Zemlya Island, which appears to be a peculiar barrier on the way of the warm Atlantic air, governs the arctic type of the climate of its area

The weather state and values of the meteorological elements over the Kara Sea area depend mainly on the intensity and interaction of the centers of the atmosphere forcing. In the autumn-fall season the atmospheric processes are governed by the effect of the Icelandic Low depression. At the beginning of the cold season north winds prevail in the northern sea, while in the southern sea the winds are unstable by direction with the speeds of 5-7 m/s. In winter the winds of southern, south-western and south-eastern directions prevail over the largest part of the sea and only in the north-east of the sea the north wind is observed. The largest number of the storms is observed in the western sea, and in winter there are 6-7 storm days a month. At the islands of Novaya Zemlya, Northern Land and the Franz-Josef Land bora is formed quite often, with the wind speed reaching 40 m/s. The maximum recorded wind speed was equal to 60 m/s. The bora lasts for several hours but in winter this phenomenon can last for two-three days. Mean monthly air temperature in winter is -20 - -28 C, the minimum - up to -48 C. In the warmer part of the year the Siberian Maximum destroys and the Low depression disappears. Due to this in spring the winds are unstable in direction and have speeds, not exceeding 5-6 m/s. In summer the local High is formed over the sea and northerly winds with the speeds of 4-5 m/s dominate. In July the air temperature on an average is equal to 5-6 C in the western part of the sea and to 1-2 C in the east and north-east. In some regions of the land coast the temperature can rise up to 18 - 20 C.

A strong continental discharge appears to be a unique feature of the Kara Sea, governing to a great extent its hydrological regime. The share of this sea [3,4] is about 55 % on an average (1290 cu.km/year) of the total discharge to all seas of the Siberian Arctic, and the discharge of the Ob' river is 450 cu.km/year of water, the Yenisey river - about 600 cu.km/year, the Pyasina river - 80 cu.km/year, the Pur and Taz rivers together - about 86 cu.km/year and other rivers - about 74 cu.km/year. Simple calculations show that only the continental discharge could fill the entire volume of the Kara Sea during a very short period of 76 years. The river run-off is distributed in time rather non-uniformly. As it can be seen from Fig.1 80 % of the continental discharge is brought to the sea during the warm period of the year from June to September. In winter the water in very small portions is discharged from only the large rivers. Depending on the prevailing winds the spreading of river waters over the Kara sea area varies from year to year. The generalization of multiyear observations allowed one to distinguish three main types [4] of the river water extensions (Fig. 2) western, eastern and fan-shaped. In general almost 40 % of the Kara Sea area experience to some extent the effect of the

land waters.

The water exchange of the Kara Sea through the straits and open sea boundaries with the adjacent oceanic and sea areas [3,5,6] is considered to be the second component of its water balance. As it is shown by calculations the inflow of the Barents Sea water through the strait Karskiye Vorota is 1240 cu. km/year, through the strait Yugorsky Shar - 400 cu. km/year and through the strait between the Novaya Zemlya and Franz-Josef Land - 17100 cu. km/year. The resulting water exchange between the Kara Sea and the Laptev Sea through the Vilkitsky strait is eastward, being 11000 cu. km/year. Thus, an approximate estimate of the water exchange with the Arctic Ocean through the boundary between the archipelagoes Franz-Josef Land and the Northern Land, following from the zero balance, indicates the resulting northward direction of the flows from the Kara Sea in the volume of 9030 cu. km/year. The presented water exchange values allow one to consider the period of the general ventilation of the Kara Sea to be less than 5 years.

The high latitudinal position of the Kara Sea governs its very weak heating in the warm part of the year [3,7,8]. At the surface the water temperature decreases from the south-west to the north-east (Fig.3), not exceeding 10 - 12 C in the southern sea. In the northern sea in the drifting ice zone the temperature in summer is only a little higher than the freezing point temperature. In the autumn-winter season the sea surface cools strongly, while in winter in the subice layer it is everywhere equal to the freezing temperature of the water of the given salinity (-1.5 - 1.7 C). Only in the troughs of St. Anna and Voronin along which the Atlantic water penetrates into the sea, the water temperature, beginning from 50-70 m levels starts to increase, reaching in the 100-200 m layer the values of 1.0-1.5 C. In summertime the thickness of the heated layer is 20 - 30 m, reaching in some years up to 50-60 m.

The main feature in the water salinity distribution appears its significant eastward and south-eastward increase from the shores of the Yamal peninsula to the Novaya Zemlya Island varying from several pro mille to more than 33 ‰ (Fig.4) [7]. In all sea regions the seasonal water salinity is well pronounced, which can be attributed to the processes of ice formation and melting and fluctuations of the river run-off. In winter the river run-off decreases with a simultaneous water salination due to the ice formation. The salinity in the upper layer, where convection occurs, increases everywhere, being in the south-western part of the sea, with the exception of the parts, directly adjacent to the river mouths, 25-30 ‰ and in the northern part and near the Novaya Zemlya Island - 33-34 ‰. In summer everywhere in the Kara Sea a well-pronounced thermo-

and halocline forms (Fig.5) at the depths from 6-8 m in shallow areas up to 20-30 m in the deep-sea areas. In shallow zones after the storm the thermo-and halocline can be completely destroyed by the wave-induced mixing.

The internal gravitational waves with an amplitude up to 10-12 m [7] and the length of several tens of kilometers, mainly of tidal frequencies often appear in the intensive layer of the pycnocline in summer (Fig.6).

In the areas of the interaction of the freshened water of the continental discharge with the saline arctic water, as well as in the zones of the melting ice edge in summer there appear strong frontal zone with the horizontal temperature gradients up to 0.5 C and salinity up to 1 ‰/km - 1.5 ‰/km [8], which on the one hand contribute to the formation of jet baroclinic currents and on the other hand attenuate a cross horizontal turbulent exchange.

On the whole the temperature and salinity distribution in the surface sea layer depends to a considerable extent, on the prevailing winds and wind-induced currents, and in the entire water column it is also governed by the water exchange and constant currents, connected with the general water circulation system in the Arctic Ocean.

The classic scheme of the water circulation in the Kara Sea [4], constructed from the observation data of the buoy and ice drift, has a prevailing cyclonic character (Fig.7). The Barents Sea water comes into the Kara Sea through the straits Karskiye Vorota and Yugorsky Shar, it reaches the Yamal coasts and shifts farther north in the form of the Yamal current. Near the northern tip of the peninsula it is strengthened by the Ob'-Yenisey run-off current, one of the branches of which goes westward through the Malygin strait, while the main flow moves northward. To the north of the Bely island the Yamal current divides. One branch turns to the north-west to the Novaya Zemlya, where combining with the water of the Western Novaya Zemlya current of the Barents Sea, which rounds the Zhelaniya Cape, it forms the Eastern Novaya Zemlya current, going to the south to the Vaigach island and which closes the cyclonic gyre in the south-western part of the sea. Near the southern tip of the Novaya Zemlya part of the waters of the Eastern Novaya Zemlya current moves through the Karskiye Vorota to the Barents Sea (Litke current). The other branch of the Yamal current, combining with the Ob'-Yenisey flow, continues motion to the north-east, forming the Western Taimyr current along the Taimyr peninsula.

The wind-induced currents, the velocities of which, as a rule, are by an order of magnitude larger than those of the constant currents [6], can significantly change the presented

circulation pattern of the Kara Sea, depending on the character of the atmospheric circulation over its area. In the coastal band with winds of any direction near the steep shore with a straight coastline the current is directed along the latter to the side, coinciding with the direction of the wind vector projection to the coastline. Near the irregular and skerry coasts the local coastal circulations with significant current velocities up to 100 cm/s appear.

The main contribution to the level oscillations of the Kara Sea is made by the storm surges and tides. The maximum values of the storm surges up to 3-3.5 m (Fig.8) [4] are observed in the bays and gulfs of the southern coast of the Kara Sea. The tidal level oscillations are not that significant, being within 30 - 80 cm.

The tides in the Kara Sea are governed mainly by the tidal wave, propagating from the Atlantic basin and through the strait Karskiye Vorota from the Barents Sea. On the whole over the sea the tides have a predominantly semi-diurnal character (Fig.9) and only near the coasts and near the Yamal and Taimyr peninsulas the tide period as affected by a complex morphometry, can differ significantly from a semi-diurnal one. In winter the ice cover influences significantly the tides: the tide value decreases a little, and the propagation of the tidal wave is delayed as compared with summer. Maximum velocities of the tidal currents are given in Fig.10, their values in open parts of the area not exceeding 20-30 cm/s, increasing in bays and gulfs of the southern coast up to 60-80 cm/s [9]. Maximum velocities of the tidal currents in syzygy can reach up to 150 cm/s. The character of the propagation of tidal wave M<sub>2</sub> is shown in Fig.12. A characteristic feature of the tides, induced by the both waves, appears to be the presence of amphidrome in the south-western part of the Kara Sea [10].

The wind-induced swell in the ice-covered seas depends not only on the speed and duration of the wind action, but also on the ice extent, as this governs the value of the wind fetch. The largest wave size is noted in the south-western Kara Sea, where with stable south-west and west winds with the speed of 14-15 m/s the waves 5-10 m high and 80-180 m long can be observed. The occurrence frequency of such waves in autumn and in winter is 15-20 % [3].

The severe climate of the high-latitudinal Kara Sea governs its complete freezing in the autumn-winter time and annual ice existence. The ice formation starts in September in the northern sea regions and in October at the south. From October to May almost the entire sea is covered with ice of a different type and age [3,4]. Fast ice occupies the coastal zone (Fig.12). It is developed non-uniformly. In the north-eastern sea the stationary



ice forms a continuous band, extending from the Bely island to the Nordenskjold archipelago and from it to the Northern Land. In summer this fast ice band breaks up and disintegrates into separate floes. They persist for a long time in the form of the Northern Land massif. In the south-western sea fast ice occupies small areas. Seaward of the stationary ice there is a zone of ice free water or young ice. This is the area of flaw leads. In the south-western sea there are Amderminskaya and Yamalskaya polynyas, and in the south of the central sea - the Ob'-Yeniseyskaya polynya. The drifting ice is common in the ice free sea areas, with the dominance of first-year ice of local origin. Their maximum thickness (in May) is 1.5-2 m. The outflow drift when the ice is outflown to the north prevails in the sea. At the south-west there is a Novaya Zemlya ice massif, which melts on site during summer. In the northern areas the ice is perennial. Here the branches of the oceanic ice massifs descend. The ice distribution in the spring-summer time depends on the prevailing wind and corresponding currents.

In the south-western Kara Sea one encounters sometimes icebergs and bergy bits, calved from the glaciers of the Franz-Josef Land and Novaya Zemlya. It appears, that most of the icebergs is of a local origin, that is, calved from the glaciers of the northern island Novaya Zemlya. Fig.13 shows a map of the position of the icebergs, observed from ships and aircraft in 1930-1950 [9]. But it follows from the map that the icebergs in the Kara Sea are mainly distributed along the coast of the Novaya Zemlya Island and are not practically encountered in the southern sea.

The hydrometeorological features of the Kara Sea, considered above, are necessary to take into account to some extent in the organization of the monitoring system and estimation of the transport, transformation and accumulation of radioactive pollution in the Kara Sea.

Analyzing the most important processes, governing the behaviour and the fate of the artificial radionuclides in the marine environment, one can make a number of the conclusions, concerning the hydrometeorological aspects of the radioactive contamination problem of the Kara Sea.

The most strong mainland discharge, as compared with the other arctic seas, formed in the extensive water catchment regions of the industrially developed areas of the Western Siberia can be one of the main sources of the pollutant input, particularly in the summertime.

The continental discharge and melting ice in summer result in the formation of a dramatic pycnocline and narrow frontal zones, which, suppressing the mixing processes can lead to the localization and increase of the radionuclide concentration. The

internal waves, appearing in the thermo-halocline layer, sharpen or attenuate the density gradients, affecting thus the vertical distribution of contaminants. That is why during the organization of the pollution monitoring it is necessary to carry out purposeful measurements of the radionuclide concentration in the pycnocline and in the areas of hydrofronts.

During observations of the radioactive pollution level it is extremely important to have information available not only on the constant currents of the Kara Sea, responsible for the long-term changes in the pollutant concentrations, but also on the structure of the quickly changing wind-induced circulation and tidal phenomena, which appear to be one of the major mechanisms of transport, transformation and accumulation of radionuclides on a short-term time scale.

The ice processes are considered to be a specific factor of the radionuclide transport and transformation in the Kara Sea. During the autumn cooling of the surface water layers the vertical convective flows, redistributing the concentration of radioactive substances are being formed. This results in the amount of the radionuclides, incoming to the forming ice, differing from the amount of radioactive substances in the surface sea layer in the summertime. Thus, the radioactive ice is being formed, the contamination of which also increases due to atmospheric precipitation in winter. The drift and melt of the polluted ice contributes to the redistribution of the concentration of the radioactive substances at water surface. Therefore the outflow of the drifting ice from the area under studies leads to a partial cleaning of this area, but to a radiation contamination of the other area, where ice melt occurs. During the ice melt the radionuclides become free and come with the melt water to the surface sea layer.

Of a particular danger for the radioactive contamination of the Kara Sea appear to be large icebergs, drifting in the areas of the radioactive waste dumping. The ice keel of the iceberg with a deep draft can destroy mechanically the dumped bodies, which will create the conditions for the radionuclide leak to the marine environment.

The study of the structure and variability of the temperature fields of the Kara Sea will allow for a more correct assessment of the biological destruction of the radioactive substances in this area.

All this, mentioned above, allows one to state that the study of the problem of the radioactive contamination of the Kara Sea is closely connected with the studies of hydrometeorological processes in its area. The collection of field data on the doses of radioactive contamination should be carried out simultaneously with the oceanographic, biological and other types of

observations of the environment.

A particular attention should be given to the further studies of the water exchange processes in the straits and at the open boundaries of the Kara Sea, water circulation and ice drift on the basis of the direct measurements and mathematical modelling. This will allow one to obtain a spatial structure and variability of the dynamic characteristics of water and ice under the influence of atmosphere forcing and features of the thermohaline state of the water masses, necessary to forecast the transfer, transformation and accumulation of the radioactive contaminants in the Kara Sea.

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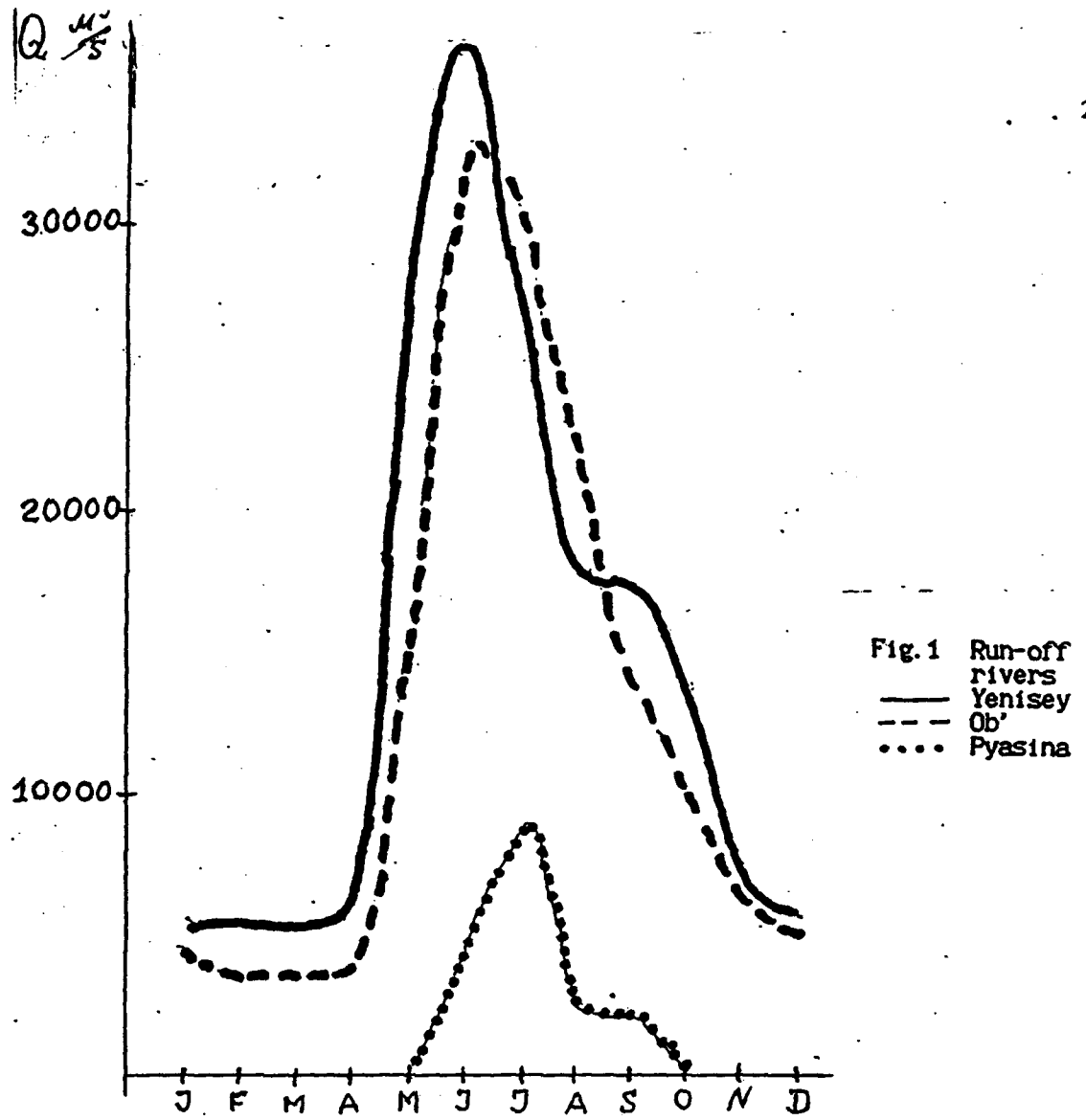


Fig. 2 Types of river water extensions (%)

a - western, b - fan-shaped, c - eastern

	70-50		80-70		90-80		>90
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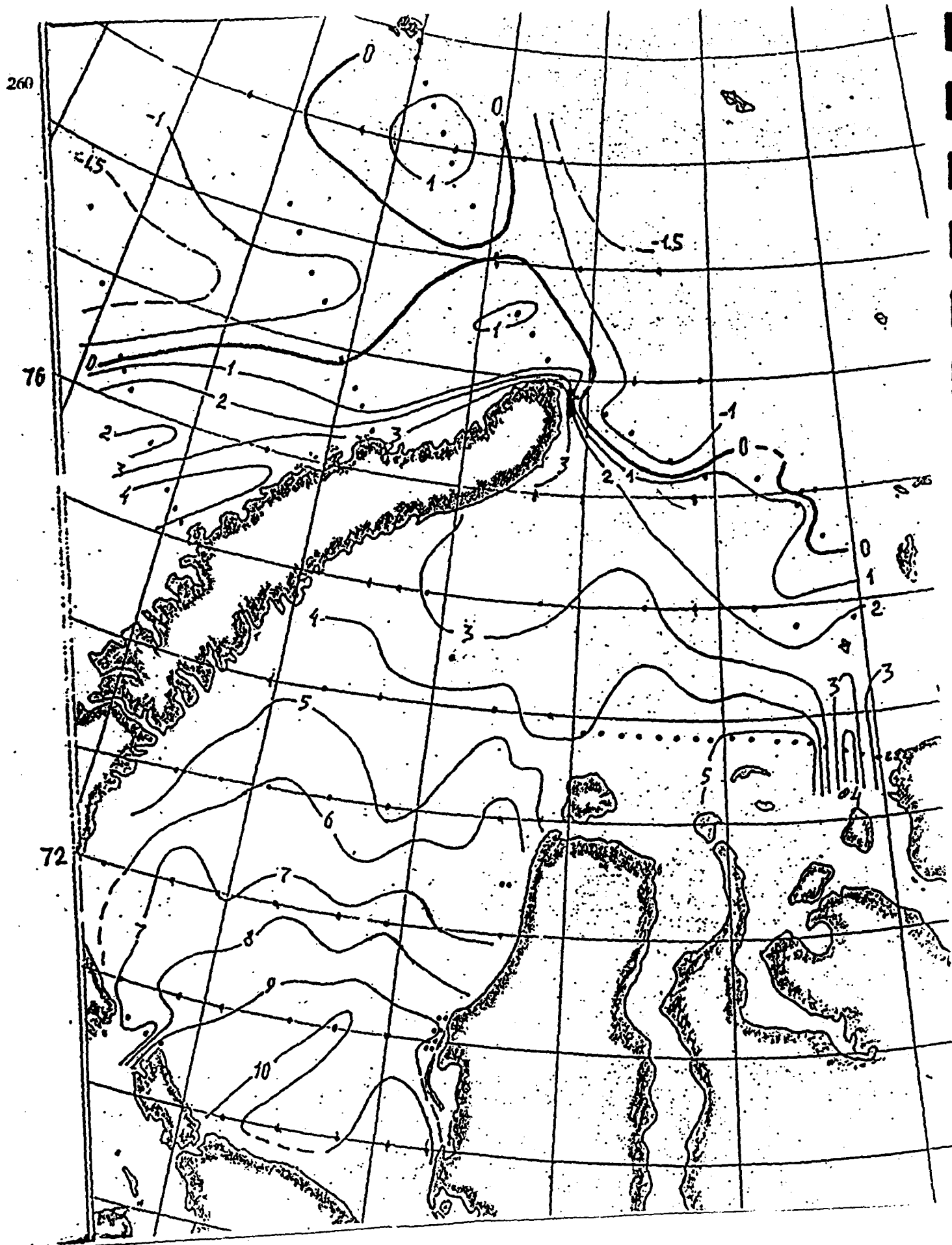


Fig.3 Water temperature at the surface

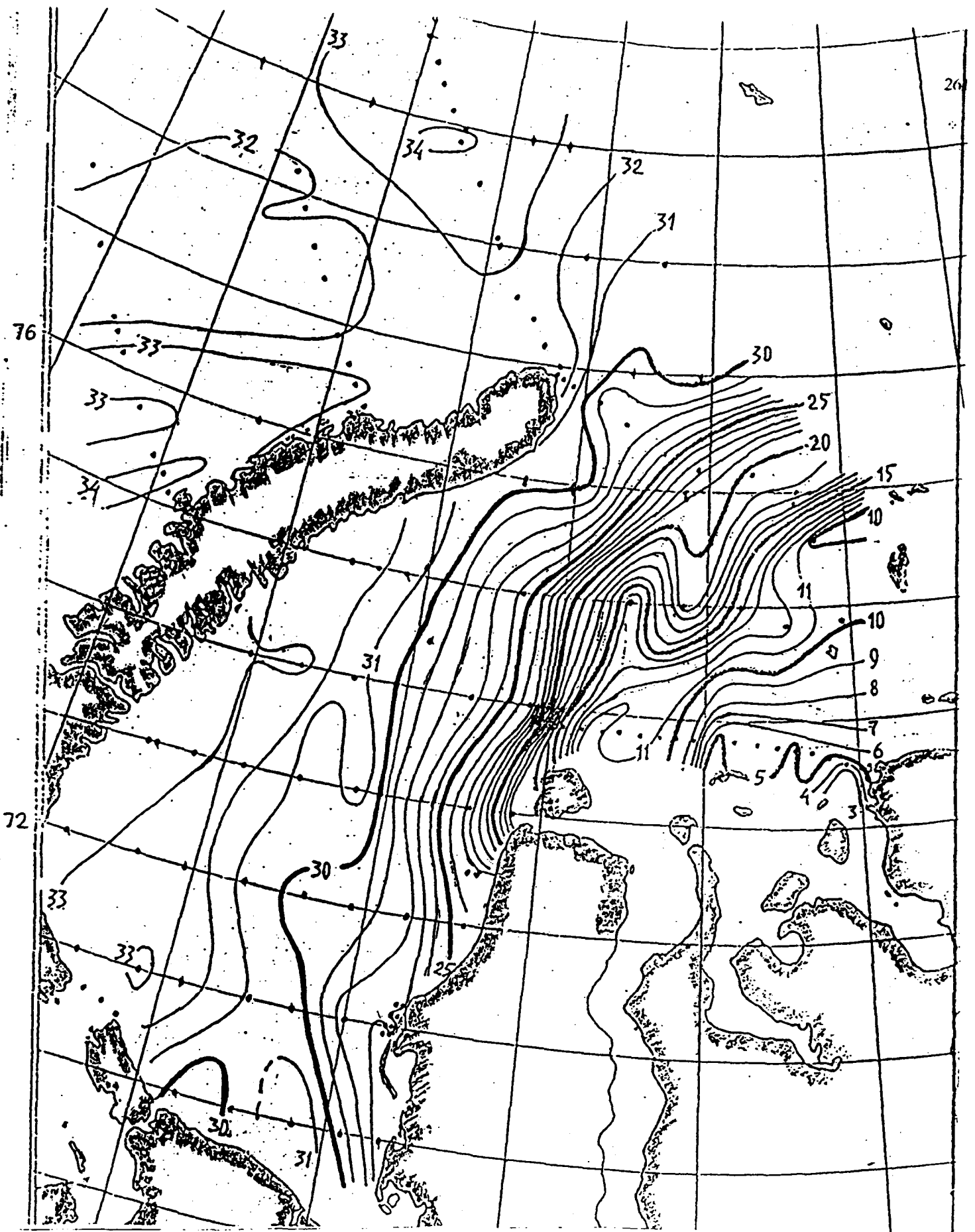


Fig. 4 Water salinity at the surface

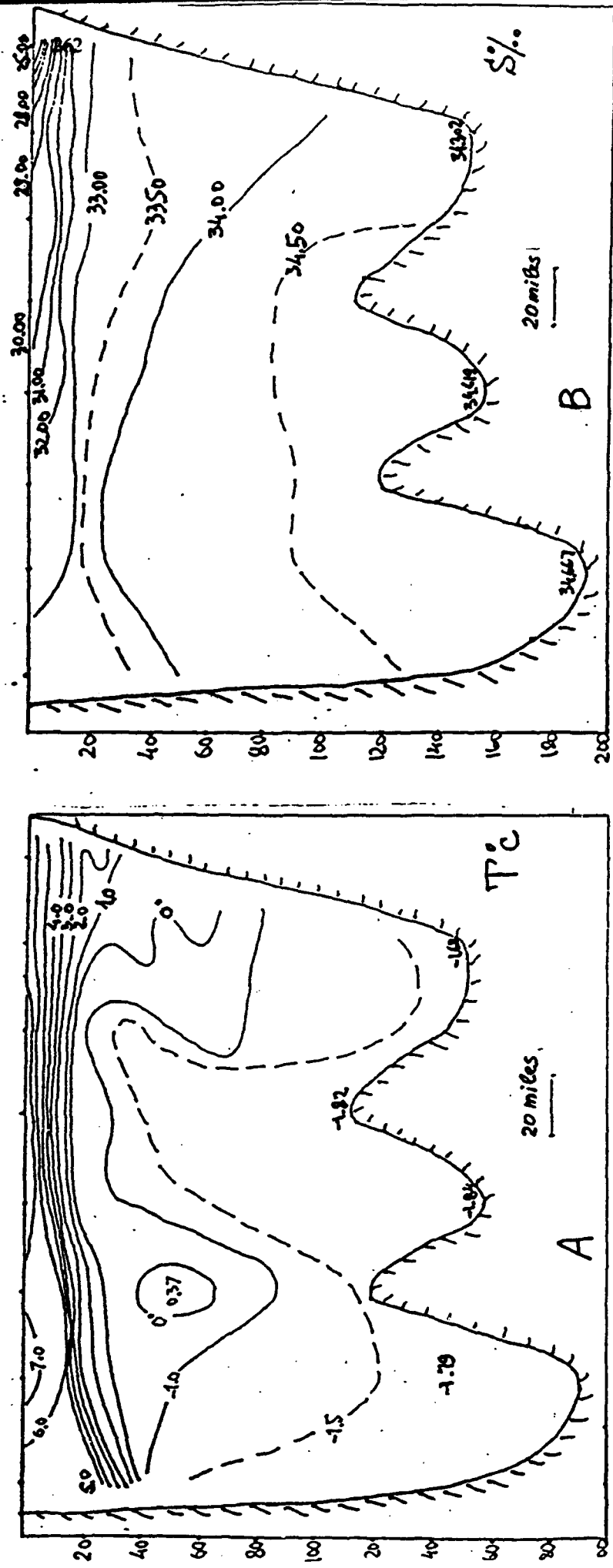


Fig.5 Temperature (A) and salinity (B) sections

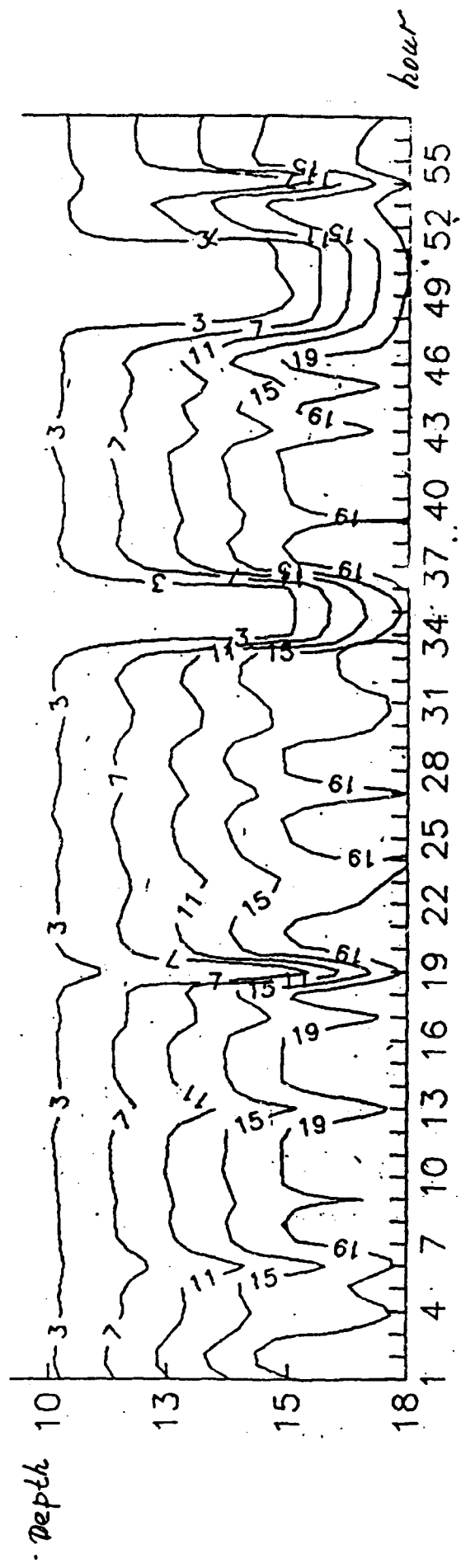


Fig.6 Time variability of location of isochalines in shallow waters

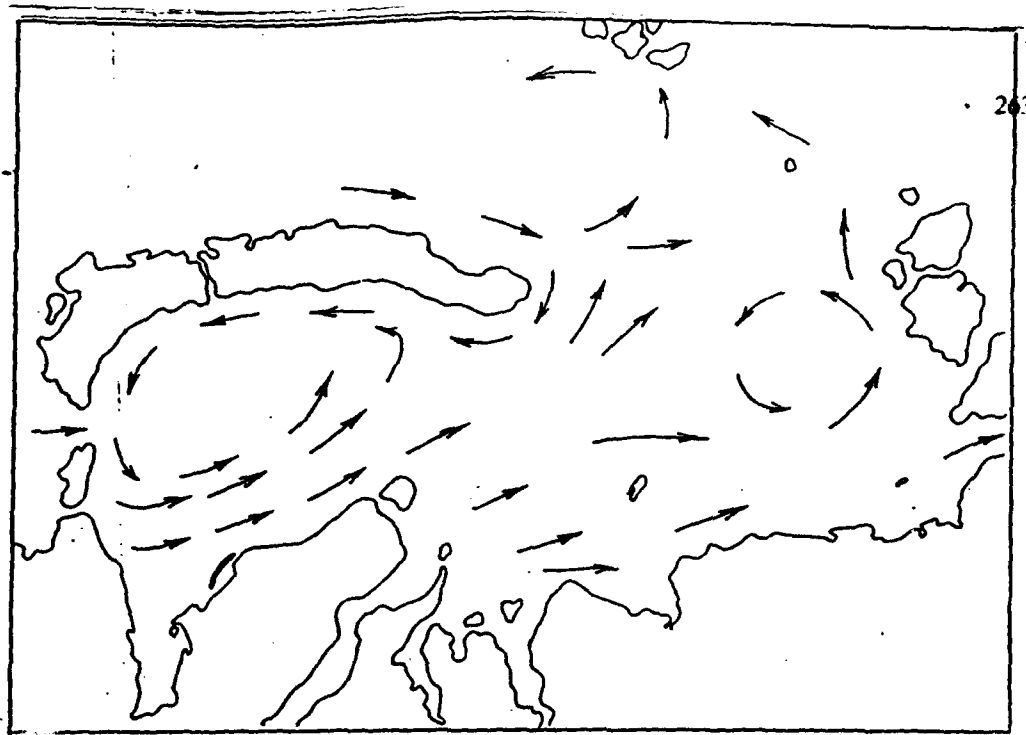


Fig.7 Scheme of the water circulation in the Kara Sea

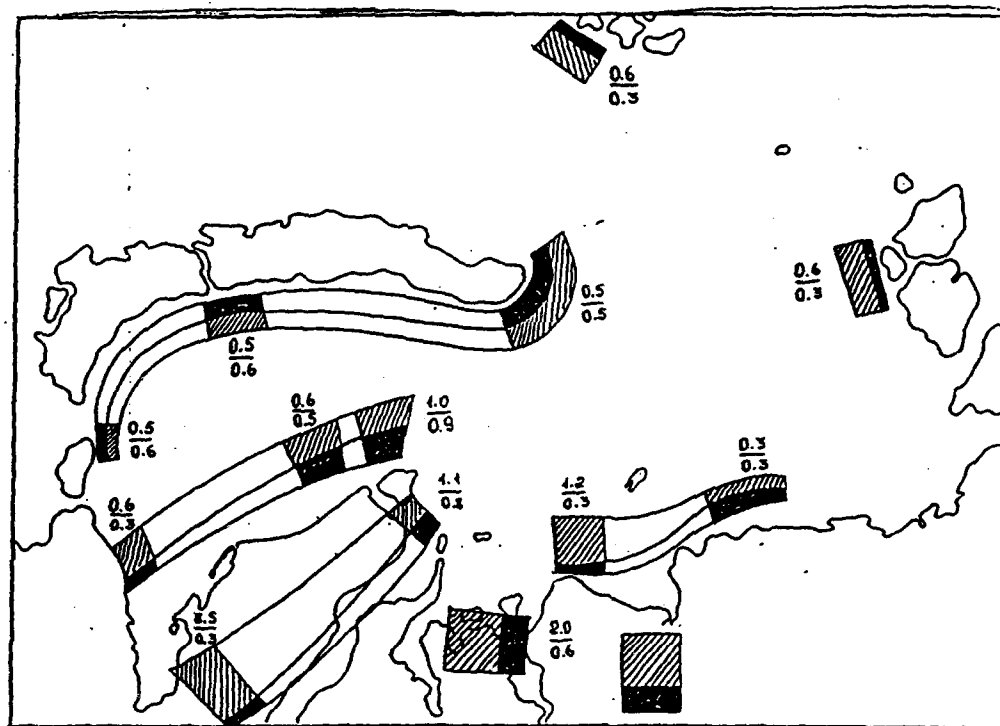


Fig.8 Values of the storm surges (numerator) and tides (denominator)



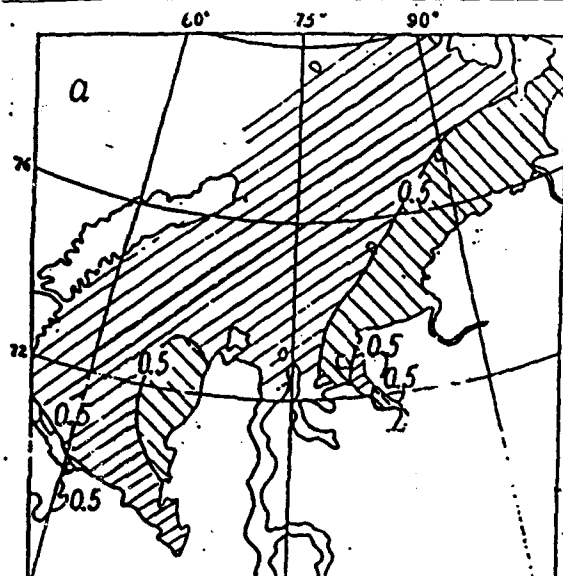


Fig. 9  
 mixed semi-diurnal tidal currents  
 semi-diurnal tidal currents

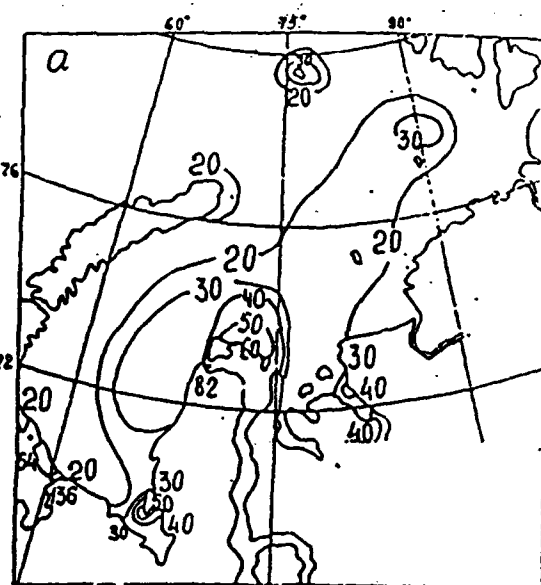


Fig. 10  
 Maximum velocities of the  
 tidal currents (cm/s)

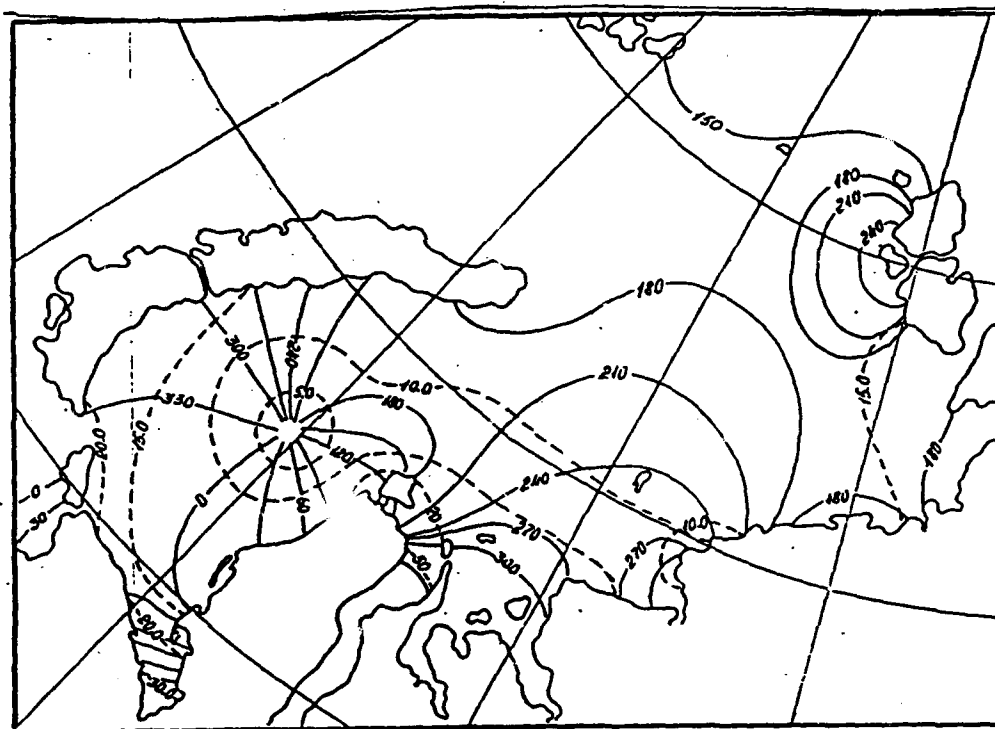


Fig. 11 M2 Tide [10]

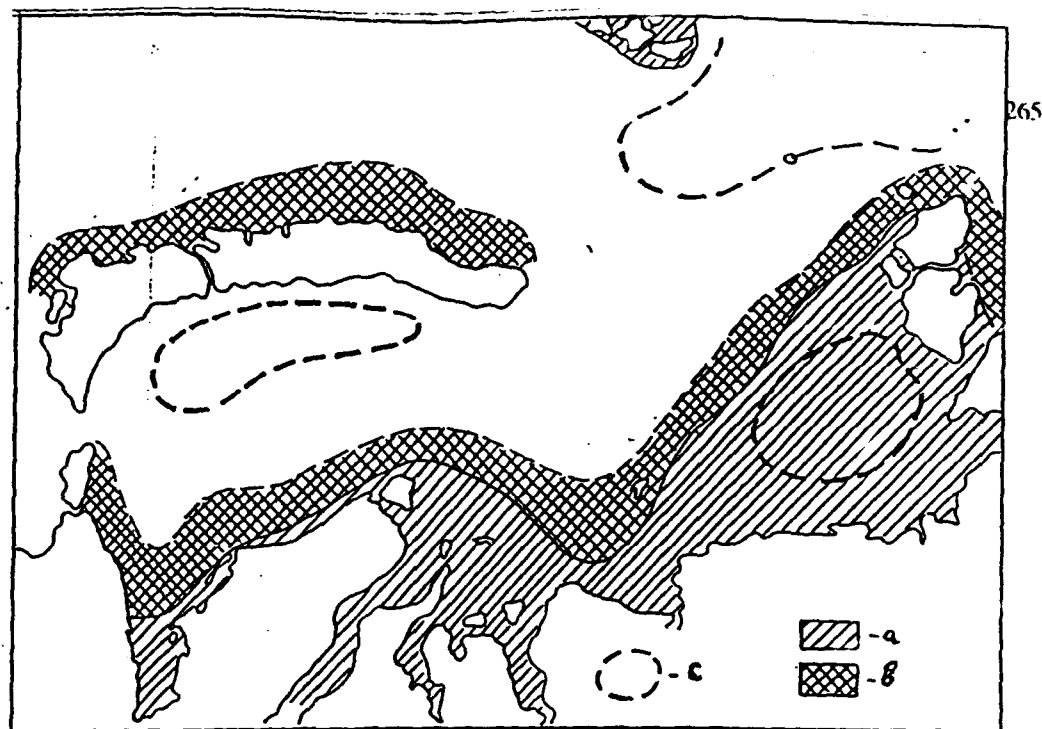


Fig. 12 Ice-cover of the Kara Sea

- a - fast ice
- b - zone of behind-fast polynyas
- c - stable ice massifs

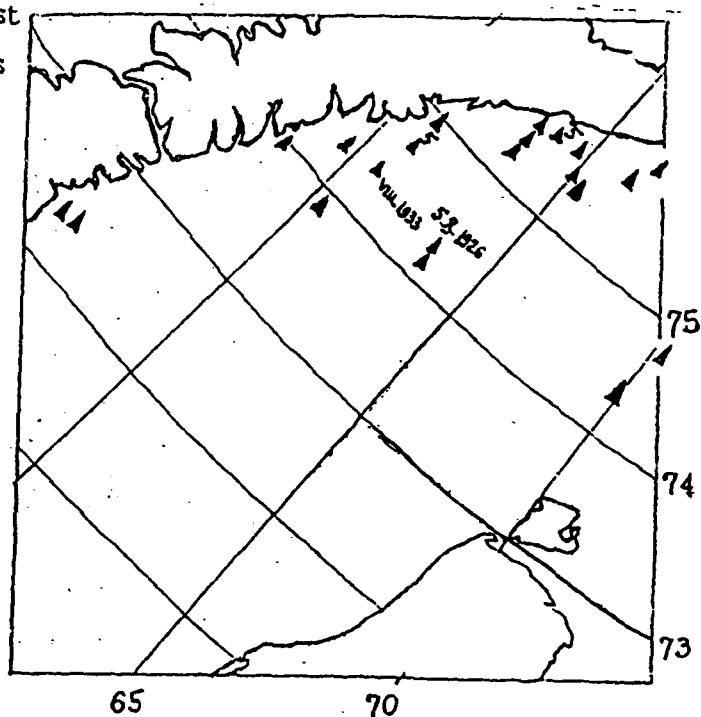


Fig. 13 Map of the position of the icebergs observed from ships and aircraft in 1930-1950



## Potential Transport of Radionuclides and Other Pollutants by Arctic Sea Ice

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### Introduction

Growing concern about Arctic pollution has led to speculation about the relative roles of the atmosphere and ocean in Arctic-wide transport of contaminants. Sea ice, one of the defining characteristics of the Arctic, is usually ignored as a potential transport medium. This paper explores the mechanisms by which ice formation and drift could influence the redistribution of contaminants, including radionuclides, in the Arctic (see also Weeks, 1993).

A major fraction of the sea ice formed during winter over shallow marginal shelves is advected and introduced into the large-scale drift patterns of the central Arctic pack. Unique to the Arctic are the large amounts of fine-grained particulate material incorporated in and transported by sea ice formed on the shelves. Estimates from expeditions across the eastern Arctic and Barents Sea (Pfirman et al., 1990; Nuernburg et al., 1993), indicate that, regionally, from 10 to 50% of the ice cover can be discolored by accumulations of lithogenic and biogenic material (figure 1). Many pollutants, including radionuclides, tend to sorb onto fine-grained particles and/or organic material (Stumm and Morgan, 1981). During several years of transport, pollutants concentrated in the oceanic surface microlayer may also be incorporated in drifting ice (Gaul, in press), while pollutants deposited from the atmosphere accumulate on the snow and ice surface.

Ice melts most extensively along the marginal ice zones. Because these are regions of intense biological activity, pollutants, often transported far from entrainment areas, are released directly into surface water where they may enter most easily into the food chain. As a result of these processes, drifting sea ice may play a role in long-range redistribution of contaminants in the Arctic.

### Shelf Processes

Recently, concern about potential releases of radioactive contaminants has focused attention on the Kara and Barents seas where nuclear materials have been dumped since 1959 (Yablokov et al., 1993). However, as highlighted by the Arctic Environmental Protection Strategy (adopted in June 1991 by the eight Arctic-rim countries), there are also circumpolar concerns about contamination from radioactive reservoirs and groundwater, pesticides, oil, heavy metals and other agricultural and industrial byproducts. Ice may incorporate these contaminants when it forms in rivers, at river mouths, and on shelf areas influenced by river discharge and dumped and leaked material (Weeks, 1993).

### River Ice

Every fall, the Siberian rivers freeze up, dramatically reducing water discharge (figure 2). River ice incorporates particles when it freezes to the bottom, as well as during anchor and frazil ice formation. Anchor ice forms when the entire water column is supercooled and ice nucleates on particles on the sea floor (see discussions of sediment incorporation mechanisms by Pfirman et al., 1990 and Reimnitz et al., 1992). A change in river temperature or turbulence, or accumulated buoyancy eventually dislodges the ice and sediment mass, rafting it to the surface. Here it is included in the overlying ice

cover. Rocks as heavy as 30 kg have been incorporated in river ice by this mechanism (Martin, 1981). Frazil ice also forms when the water column is supercooled. Particles adhere to the surface of the ice crystals, and are incorporated in the growing ice cover. Ice formed in this way appears turbid, with patchy discolorations due to entrained material (Reimnitz et al., 1992). In addition, wind-blown dust and dirty snow accumulate locally on the surface of the frozen rivers. As a result of these processes, river ice often contains large amounts of sediment (Zubov, 1943).

Spring break-up of river ice is a violent event. Ice dams form, then are destroyed with massive discharges of water and ice. River beds are gouged by tumbling ice and eroded by torrential water flow. Polluted sediments and waste containers on the river bed may be disturbed by these events. In the lower reaches of the Yenisey and Lena rivers, the water level rises more than 20 m (Antonov, 1970). Siberian rivers discharging into the Kara, Laptev and East Siberian seas have a huge combined drainage area of  $9 \times 10^6 \text{ km}^2$  extending far to the south (Shiklomanov, 1993), encompassing many industrial and agricultural regions. The river Ob and its tributaries originate as far south as  $45^\circ\text{N}$  (Futsaeter et al., 1991). Pollutants accumulated in watersheds during the winter may contaminate river flow during spring and summer.

River ice discharged at the river mouth most likely melts and deposits its particle load in the nearshore zone (Reimnitz and Bruder, 1972). This happens because the main period of discharge is during June (figure 2) when the Arctic summer begins and ice in the marginal seas is just beginning to melt. If river break-up occurs before there is much melting of the shorefast ice, river water may flow out over the ice, depositing its sediment load on the surface (Reimnitz and Bruder, 1972). Because of the low albedo of this ice, it melts rapidly in early summer. In part because of the discharge of river water and ice, river estuaries are usually centers of initial ice melting, (Zubov, 1943). The river water is warmer than the  $< 0^\circ$  shelf water, and rapidly melts sea ice in a region near the river mouth (Antonov, 1970). Also, particle-laden river ice has a lower albedo, and therefore melts more quickly than cleaner ice.

Another factor contributing to retention of river ice in the nearshore zone is the extensive fast ice cover that forms along the Siberian margin each winter (figure 3). This ice is anchored to the coast and shallow banks, forming a barrier to offshore transport for any floating material (i.e. ice) that is released behind it. Particle-laden river ice, as well as ice influenced by bluff slumping and bottom adfreezing is retained near shore in this way.

Some river and shorefast ice undoubtedly does make its way past these barriers, survives transport across the shelf and is incorporated in the large-scale drift of the Arctic ice pack. A distinguishing characteristic of river ice is that it has low  $\delta^{18}\text{O}$  values: Ob River water is about  $-16.2\text{‰}$  and Yenisey River water is about  $-17.9\text{‰}$  (Brezgunov et al., 1982), and Lena River water is approximately  $-21\text{‰}$ , while Eurasian Arctic surface water tends to be between 0 to  $-2.9\text{‰}$  (Grabitz et al., in prep.). At present, there is not enough information on this transport pathway to actually quantify the amount of river ice that is incorporated in the Arctic ice pack. However, according to Zubov (1943) "inasmuch as this ice is almost completely destroyed during the course of a polar summer its [river ice's] role in the ice regime of the seas is extremely insignificant."

### *Sea Ice*

Ice crystals growing in the sea exclude salt, resulting in ice with a lower bulk salinity than the water from which it forms. In this way, soluble salts (Weeks and Ackley, 1986) and some contaminants that are dissolved in the water column are likely to be excluded from the ice (Weeks, 1993). However, Arctic ice forming over shallow Siberian seas often includes sediments and organic material. Some pollutants tend to sorb onto this material, and many radionuclides are very particle-reactive (Stumm and Morgan, 1981). This effect is enhanced at colder temperatures. Tanabe and Tatsukawa (1983)

observed that a high percentage of DDT and PCBs were adsorbed to suspended particles at high latitudes.

Sea ice formed over the Siberian shelves incorporates particles predominantly during suspension freezing and frazil ice formation, but also as a result of anchor ice rafting (Reimnitz et al., 1992). Most particle-laden ice appears to form in water depths < 50 m (Reimnitz et al., 1993). This is because the energy needed to resuspend sea floor sediments through the water column increases with depth. Also, anchor ice growth requires that the entire water column is supercooled. While particles entrained during frazil ice formation are silt-sized or smaller, boulders may be rafted from the sea floor by anchor ice.

Processes associated with frazil ice formation, tend to cause elevated levels of suspended particulate matter in the water column (Kempema et al., 1989). Combined with wave activity and scavenging by ice crystals, the initial ice cover may become enriched in particulate matter relative to normal concentrations observed in the underlying ocean water (Ackley, 1982, Garrison et al., 1983, Reimnitz et al., 1990, Shen and Ackermann, 1990). These mechanisms could also contribute to the elevated levels of organic material observed in Arctic sea ice. Concentrations of suspended organic carbon may be two orders of magnitude higher than sea water (175 to 560  $\mu\text{g/l}$  compared with 25-45  $\mu\text{g/l}$  in June to August and approximately 5  $\mu\text{g/l}$  during the remainder of the year, Mel'nikov and Pavlov, 1978). According to these authors, the elevated levels appear to result from infreezing of organic material during ice formation on marginal Arctic seas which have higher biological productivity than the central Arctic Basin. When ice melts in summer, some suspended organic material is contributed to the surface sea water (Mel'nikov and Pavlov, 1978). Dissolved organic carbon may also be enriched within the ice cover due to adsorption. Because many pollutants are preferentially associated with fine-grained sediment (clay) and/or organic material, incorporation of such material on the shelves provides a process for enriching sea ice formed there with contaminants. Therefore, while sea ice without incorporated sediments or organic material probably has less of a dissolved pollutant load than the water from which it grew due to exclusion of salts and other impurities (Weeks, 1993), sediment/organic-rich sea ice would tend to have elevated contaminant loads.

In winter and spring, the Arctic atmosphere contains high levels of pollutants from Eurasia, known as Arctic haze. Sea ice acts as a lid, or sediment trap, on the surface of the Arctic Ocean. Each year the floe drifts, pollutants and other materials are deposited on its surface from the atmosphere in snow, rain and as dry deposition. Heavy metals accumulating in the snow cover of central Arctic sea ice can reach values that are characteristic of snow deposits on sea ice near Siberian industrial areas (Melnikov, 1991). Pollutants deposited on sea ice by atmospheric transport could be incorporated in the ice surface when meltwater refreezes (figure 4), and could also be added to the ice underside when meltwater runs off and refreezes. During drift, pollutants concentrated in the oceanic surface microlayer may also be incorporated in the ice (Gaul, in press). As noted by Zubov (1943), "in this manner the sea ice sucks in nutritive matter from the atmosphere on the one hand, and on the other the turbidity and organic matter from the entire water layer which is involved in the mixing process."

#### Central Arctic Pack Ice

Melnikov (1991) considers processes related to sea ice formation, drift and ice and snow melting to be one of the main factors governing surface ocean metal concentrations in the Arctic. Similarly, Pavlov and Volkov (1993) and Pavlov (1993) note that sea ice formed in the Kara Sea will incorporate radionuclides from the sea as well as those deposited from the atmosphere and release these pollutants when the ice melts. This means that pollutants accumulated throughout the fall and winter are released during spring snow melt and summer sea ice decay. The melt period coincides with the spring bloom of biological activity, increasing the potential for biological uptake of pollutants (Melnikov, 1991). This process can influence shelf ecosystems when first year ice and accumulated snow melt. It is also important for ice that exits the shelf and drifts within the central Arctic pack ice.

Much of the ice within the central Arctic ice cover formed initially on the marginal seas (figure 5). In particular, the wide, shallow Siberian shelf seas are a major source. The Laptev Sea appears to discharge the most ice each year to the central Arctic, followed by the Kara Sea and, to a lesser degree, the Barents Sea (Zacharov, 1976). The East Siberian Sea imports some ice each year from the Arctic Basin. Combined action of winds and currents cause large seasonal variations in the transport of ice, both onto and off of the shelves. Both the Barents and Kara seas export ice to the Arctic Basin in winter and import ice in summer. In order to assess potential pollutant transport, detailed data is required on both import and export of ice that is formed in regions where it may entrain contaminants.

Most of the sea ice exported to the central Arctic from the shelves, forms along the polynya at the edge of the fast ice (figure 3) and over the outer parts of the shelf seas. In the Eurasian Arctic, drifting ice is incorporated into the Transpolar Drift Stream, moving west toward Fram Strait (figure 6). Transport from the Laptev Sea to Fram Strait typically takes about 3 years (figure 7). Ice in the western Arctic, contributed from the Beaufort, Chukchi, and East Siberian seas, is incorporated in the anti-cyclonic Beaufort Gyre. Ice may circulate in this gyre for 5 years (Thorndike, 1986).

During transport, the original floe is modified substantially (see Pfirman, et al., 1990). Each summer, all of the snow and between 32 and 70 g/cm<sup>2</sup> of ice melts off the surface (average is 40 g/cm<sup>2</sup>, Hanson, 1965). Melting snow runs off the floe, percolates into the floe surface, accumulates in melt ponds, and may refreeze on the ice underside, redistributing some contaminants originally located on the ice surface. Some dissolved and particle-associated pollutants are also lost to the water column with the meltwater. Each winter, more ice is added to the underside of the floe (figure 8). As a result, while the floe thickens, the original ice, perhaps 1.5m thick, eventually melts away. Particles, distributed within the ice during formation on the shelves, eventually melt out and often accumulate on the ice surface. Zubov (1943) states that every particle frozen into the ice from below will appear on the surface in 2 to 3 years. Because of their darker color, particles absorb more solar energy and melt the ice around them, forming accumulations in pits, called cryoconite holes (figure 1). This pitting process is important because it concentrates the particles as well as retains much—but not all—of the particle load on the ice surface, even when the ice meltwater runs off the floe or the floe is tipped or submerged during a rafting event. Therefore, drifting ice that originally contained dispersed contaminant-laden particles would tend to form concentrated accumulations at the surface as time progresses (figure 8). Maximum surface concentrations from ice melt could be expected within about 3 years, if about 50 cm of ice melt off the ice surface each year and the original ice is 1.5 m thick. After this time, no additional contaminants are added to the surface from the melting ice, although atmospheric deposition will continue.

### Pollutant Release in Marginal Ice Zones

#### *Fram Strait and East Greenland*

The main exit for Arctic sea ice is through Fram Strait. Each year about 2,600 km<sup>3</sup> of sea ice (representing about 1 million km<sup>2</sup>) is exported through this region in the East Greenland Current (Kvambekk and Vinje, 1993; note earlier estimates had placed the total volume closer to 5,000 km<sup>3</sup>, Vinje and Finnekåsa, 1986). For comparison, this volume of ice is approximately equal to the volume of Siberian river discharge (2,525 km<sup>3</sup>, Zubov, 1943; 2,340 km<sup>3</sup>, Melnikov, 1991). Between 50 to 85% of the ice discharge consists of multiyear and second year ice (Vinje and Finnekåsa, 1986), which potentially contains accumulated pollutants. The marginal ice zone extends southward, along the eastern slope off Greenland (figure 6). In winter, ice also continues around the southern tip of Greenland and extends up into Baffin Bay.

Because of formation of cryoconites on the floe surface, much of its particle load will be released when the entire floe disintegrates during melting. Sediment traps on moorings deployed across Fram Strait show that traps located in the marginal ice zone, accumulated much more ice-rafted debris than

traps located underneath the ice stream to the west, where there was a persistent ice cover (Hebbeln and Wefer, 1991). Therefore, release of contaminant-laden particles is expected to be greatest along the marginal ice zone.

An important point is that the surface accumulation of particles/pollutants will be released to the sea surface when the ice melts. This concept is emphasized by Pavlov and Volkov (1993) and Pavlov (1993) who conclude that drifting ice contaminated with pollutants will "partially clean" the area of pollutant incorporation, but will also lead to contamination of surface sea waters in the region where ice melt occurs. Unfortunately, most melting and therefore particle/pollutant release occurs in the marginal ice zone, where there is a great amount of biological activity in the surface waters. Here fauna associated with the ice form an important link in the food chain between primary producers and fish, sea birds and mammals (Futsaeter et al., 1991). If ice fauna are contaminated by pollutants carried by sea ice, they will contribute these pollutants to the Arctic ecosystem.

A mitigating factor that may be important is that during transport, freeze/thaw cycling tends to aggregate particles into pellets on the ice surface (Barnes and Reimnitz, 1974; Barnes et al., 1990; Goldschmidt et al., 1992). These pellets were also observed in sediment traps located under the ice (Berner and Wefer, 1990). Pelletization results in increased sedimentation rates of the particles and their pollutant load, moving them out of the surface layer much more rapidly than if they were released as single particles. Very little data is available on particle release from sea ice to determine the relative importance of particle aggregation.

#### *Barents Sea*

The Barents Sea is a vulnerable area with extensive fishing activity, a large amount of ice melting and close proximity to pollutant sources in the eastern Barents and Kara seas. It is one of the most highly productive seas in the world. Fishing focuses on the marginal ice zone, where biologic activity is concentrated. Although much of the sea ice in the Barents Sea forms locally, it also receives ice from the Kara Sea and the Arctic Ocean. More than 40% of the ice may be multiyear (Loeng and Vinje, 1979). According to Vinje (1985), who assumed an average ice thickness of 2m, the Barents Sea imports 37 km<sup>3</sup> from the Arctic Ocean, and exports 72 km<sup>3</sup>. Most of the import is between the months of April and June. Ice import from the Kara Sea in winter is an order of magnitude greater (629 km<sup>3</sup>; Vinje, 1987). During June, July, August and December, the flow reverses and 72 km<sup>3</sup> of ice are exported to the Kara Sea. Ice coring in the western Barents Sea in May 1989, confirmed that much of the sea ice analyzed was imported from elsewhere (Pfirman et al., in prep.). This conclusion is based on analysis of  $\delta^{18}\text{O}$  values as well as physical characteristics of the ice. Sea ice formed in the Barents Sea should generally have values > 0‰, and most ice sampled was < 0‰ (figure 4). Pfirman et al. (1993) suggest that some of the ice could have formed in the Kara Sea in a region that was affected by discharge from the Ob and Yenisey rivers. Abelmann (1992), based on analysis of sea ice diatom assemblages, also concluded that ice sampled in the western Barents Sea in 1987 probably originated in parts of the Kara or Barents sea that were influenced by rivers.

#### *Other Marginal Seas*

The large-scale circulation of ice in the Arctic generally results in export of ice from the shelf seas, transport over the central basin and discharge through Fram Strait. However, modeling of ice motion (Colony and Thorndike, 1985) indicates that ice that melts in shelf regions, does contain some contribution from the central Arctic (figure 9). This is important to remember when considering deposition of pollutants from the atmosphere. For example, sea ice melting in the Beaufort Sea is likely to contain some ice from the north, which may have accumulated deposits of Eurasian atmospheric pollutants.



## Sea Ice Contaminant Data

Pollutants of primary concern are organochlorines, heavy metals, radionuclides and oil. Data on actual pollutant levels in Arctic sea ice are sparse. We have not been able to locate any information on radionuclide concentrations in sea ice. The most comprehensive information available to date on sea ice contaminants is from Melnikov and Vlasov, 1997 (figure 10). These data, reported as average values for various Siberian seas in 1990, indicate that the ice generally has intermediate levels of contaminants compared with the snow above and the water below. Total PCB concentrations in the ice typically were equal to or greater than those in the overlying snow and underlying water (ice: 1,500 to 2,500 pg/l, sea water: 1,000 pg/l, snow: 1,500 pg/l). Dethleff et al. (1993) found values ranging between 100 to 3,000 pg/l in sea ice and water of the Laptev Sea.

Near Fram Strait, Gaul (in press) observed concentrations in drifting, porous sea ice of 1,200 to 1,600 pg/l for DDT and 15,500 to 20,300 pg/l for PCB 138 at a station located just north of Svalbard at 80°48'N and 8°46'E (figure 10). Gaul notes that the ice contained (shrimp?) "fecal pellets" (probably actually sediment pellets, Gaul, pers.com. 1993) which may explain the "amazing" amounts of DDT and PCB 138. Two stations further south in the Greenland Sea had lower levels of organochlorines and did not contain notable amounts of particulates. Levels of DDT and PCBs in these samples were elevated in ice relative to surface sea water (figure 10), but in one case the amount of HCH was higher in the water than the ice (Gaul, in press). The ice may have accumulated the pollutants during formation and/or drift. Analyses of origin, drift path and age of the ice would have to be carried out to determine the reasons for the elevated concentrations.

In contrast, first year sea ice with low particle content at the Canadian Ice Island contained generally low levels of organochlorines (Hargrave et al., 1988). Concentrations of HCH in ice meltwater were lower by up to an order of magnitude than those in sea water (figure 10). Interestingly, dieldrin concentrations in the ice were three times higher than in sea water. No detectable concentrations of HCB, chlordane, isomers of DDT and DDE and congeners of PCBs were observed in the ice samples, however, detectable amounts were observed in particulate matter collected from the bottom 10 cm of the ice (Muir et al, 1992). Low values of contaminants would be expected to be found in this ice that formed from fairly clean water, had a generally low particle content, and had not drifted for very long and so did not have much of a chance to pick up pollutants from the surface microlayer.

Pb, Fe, Cu, and Cd typically are elevated in sea ice/snow compared with surface ocean water (Melnikov, 1991), perhaps due to atmospheric deposition and infreezing of particulate matter. Underice observations indicate that starting in March, Pb, Fe and Cu are released from the ice, apparently due to brine migration, resulting in concentrations in the surface water that are 2 to 3 times higher than the initial values (Pb increased from .1 µg/kg to >.2 µg/kg, Fe from <.5 µg/kg to >1.0 µg/kg, and Cu from <.1 µg/kg to >.2 µg/kg, Melnikov, 1991). Campbell and Yeats (1982) similarly concluded that ice melting contributed Fe, Cu, and Cd to surface waters in northwest Baffin Bay. In their study, sea ice, with notably high particulate concentrations (4.75 mg/l), collected off Bylot Island yielded concentrations of these metals in excess of the levels observed in surface waters (Ice: Fe 25.28 to 59.90 µg/l, Cu 8.22 to 7.29 µg/l, Cd 0.31 mg/l, Water: Fe .60 to 3.07 µg/l, Cu .18 to .60 mg/l, Cd .020 to .075 µg/l).

## Ice-related Processes

### *Shelf Brines*

Another way that sea ice may influence pollutant distribution is by the transport of shelf brines associated with ice formation. Salt excluded during crystal growth is added to the underlying water column (figure 11). Highly saline water is formed in this way in many shelf regions, particularly where the water depth is shallow—allowing surface to bottom convection—and where large amounts of ice are formed: e.g the polynya along the Siberian fast ice zone (figure 3). According to Pavlov (1993),

such vertical convection associated with ice formation redistributes the concentration of radioactive substances.

Shelf brines run off the banks, and flow into neighboring depressions, probably carrying sediments and associated contaminants with them along the way. On shelves with complicated topography, such as the Barents and Kara seas, it is likely that the brines will accumulate in enclosed depressions. When brines accumulate sufficiently to overtop and overflow shelf depressions, and where shelf troughs extend across the shelf to the edge, the brines will sink to their density level (figure 11) and then flow along the slope towards the east.

### *Ice Gouging*

Another process that effects materials on the sea floor, is gouging by sea ice pressure ridges and icebergs (Weeks, 1993). Sea ice pressure ridges on the Siberian shelves often have a draft of 25 m (Zubov, 1943) and have been documented in the Eurasian Arctic extending down to 26-43 m (Wadhams, 1986). While recent iceberg gouging in the Barents Sea affects the sea floor down to 120 to 130 m water depth (Elverhøi et al., 1989) due to calving from glaciers on Nordaustlandet (Svalbard) and Frans Josef Land, most icebergs observed today are generally less than 100 m thick (Vinje, 1985). Some glaciers on the northern island of Novaya Zemlya calve into the sea. The icebergs documented by Pavlov (1993) to the east of this region, may come from here although according to Zubov (1943), most of the Novaya Zemlya icebergs are trapped in shallow fjords. Severnaya Zemlya is also a source of icebergs to the Siberian seas. Icebergs in the Laptev Sea reportedly ground in water depths up to 183 m (600 ft: Kovacs, 1972). While ice gouges may penetrate more than 5 m into the sea floor, in the Barents Sea typical plough mark relief is 2 to 5 m and width is 10 to 50 m (Elverhøi et al., 1989).

The most important effects of ice gouging are most likely the damage that it can do to containers on the sea floor (Weeks, 1993) and the release of contaminants to the water column when sediments are physically reworked by the ice. Materials dumped in the shallow fjords of Novaya Zemlya (Yablokov et al., 1993) could be affected by these processes. Sediment transport via adhering and adfreezing to the ice mass is probably not as important.

### *Conclusions*

Because many dissolved pollutants are excluded with salt during the freezing process (Weeks, 1993) sea ice without incorporated sediments or organic material may be less contaminated than the water from which it forms. However, much of the ice formed in shallow regions (< 50m) of the Siberian seas entrains resuspended shelf sediments and organic material, and may therefore incorporate associated contaminants. Some of this ice is exported from the shelf and transported over thousands of kilometers to the Greenland and Barents seas.

Due to surface melting during transport, particle concentration typically increases in the upper portion of the ice cover, often forming a slurry of mixed sediment and organic material after several years of transport. Pollutants, originally distributed throughout the ice, may become concentrated during drift on the ice surface in this way. Although some pollutants are lost in meltwater runoff, contaminants are also added from atmospheric deposition of Arctic haze, as well as from the surface ocean microlayer.

Most particles, and associated contaminants, are probably released at the sea surface along the marginal ice zone. Because of the intense biological activity on this region, pollutants released here can easily enter the food chain. Therefore, the biological communities most at risk from long-range pollutant transport by sea ice are those along the marginal ice zones of the Barents and Greenland seas, as well as the Iceland sea and the west coast of Greenland and Baffin Bay (Weeks, 1993).

This process of retaining particle/pollutant concentration during transport and releasing it at the sea surface over a thousand kilometers away, makes sea ice a unique and potentially important transport mechanism. Transport of pollutants by wind and oceanic processes tend to result in dispersed and reduced concentrations downstream. Also, due to brine formation and sedimentation of pollutants on the shelves, oceanic processes in the Arctic tend to move contaminants downward, out of the surface water layer. The opposite is true in the case of contaminant transport by sea ice.

#### Future Work

As documented in this paper, transport by sea ice is unique, but nearly no data are available to permit assessment of its relative contribution to pollutant redistribution in the Arctic. Future studies should focus on the entrainment, transport and release of contaminants by sea ice. For example, how are different pollutants incorporated into sea ice growing under various conditions, e.g. with and without entrainment of sediment or organic matter? Where does river ice melt? Are there regions on the shelves where pollutant incorporation is particularly active? Where and under what conditions are polluted river and/or sea ice transported off the shelves? How are pollutants transformed during transport? What is the relative importance of atmospheric deposition vs. infreezing in contributing to the pollutant load of sea ice? How important is seasonal contaminant release during transport? What happens to pollutants released when an entire floe disintegrates? Are organisms associated with the ice contaminated? How important is particle aggregation?

Because many pollutants are likely to be associated with fine-grained sediments and organic material, prime candidates for assessment of long-range pollutant transport are "dirty" sea ice floes in the Barents Sea and east Greenland marginal sea ice zones. Investigation of ice exchange between the Barents and Kara seas should also have high priority because the Kara Sea contributes such a large amount of ice to the Barents Sea and it is influenced by the polluted Ob and Yenisey rivers. Investigations of pollutant load should be coupled with analyses of ice physical characteristics, sediment and organic carbon content, and tracers, such as  $\delta^{18}\text{O}$ , particle composition (clay and heavy minerals, diatom assemblages), and organochlorine ratios. By carrying out such integrated investigations, pollutant incorporation mechanisms can be reconstructed as well as the origin, development, and transport history of individual floes.

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Figure 1. Photograph showing surface sediment distribution on multiyear sea ice (by I. Wollenburg, from Goldschmidt et al., 1992). Sediment often is formed into pellets and accumulates in pits on the ice surface, called cryoconite holes.

### Siberian River Runoff

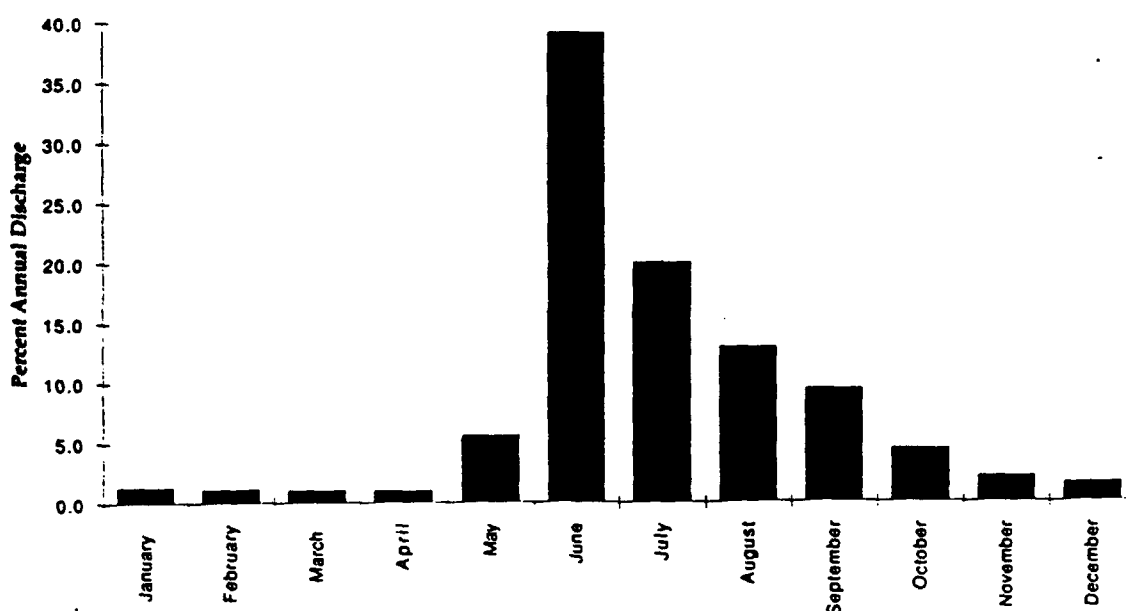


Figure 2. Total inflow of the Siberian rivers to the Arctic Ocean (% per month) for characteristic years (Shiklomanov, 1993). Note the dramatic increase in river discharge in June.

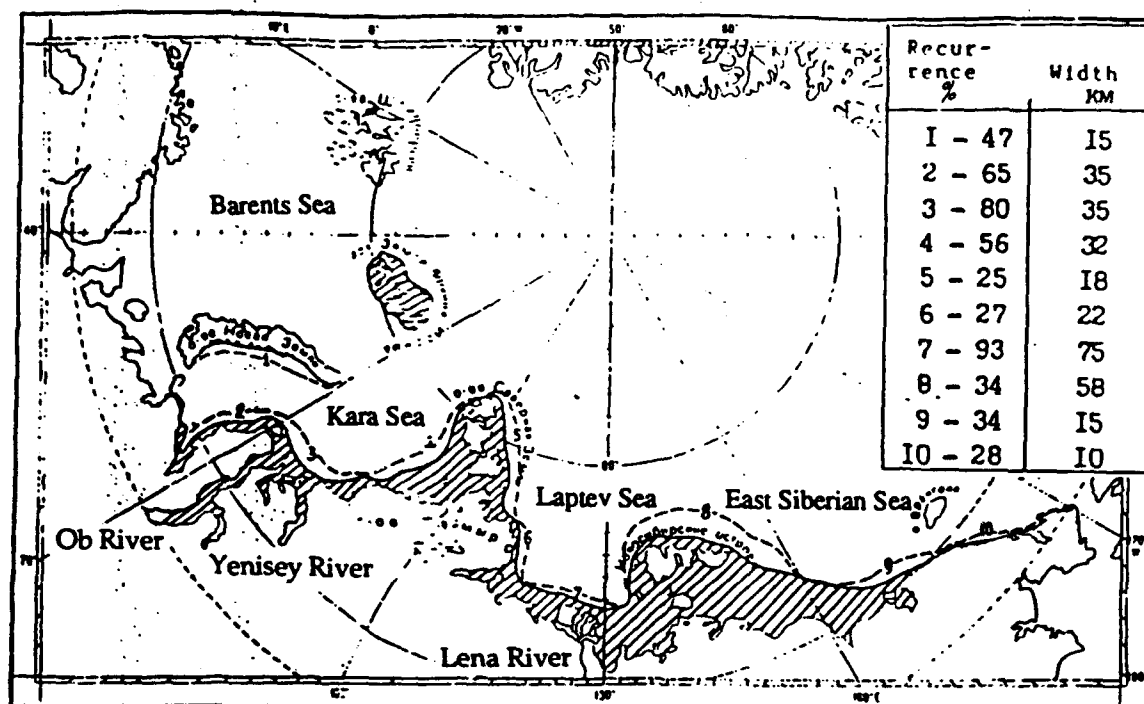


Figure 3. Extent of shorefast ice and adjacent polynya along the Siberian shelf seas during May to June, the period of maximum development of the ice cover (Buzov, 1991).

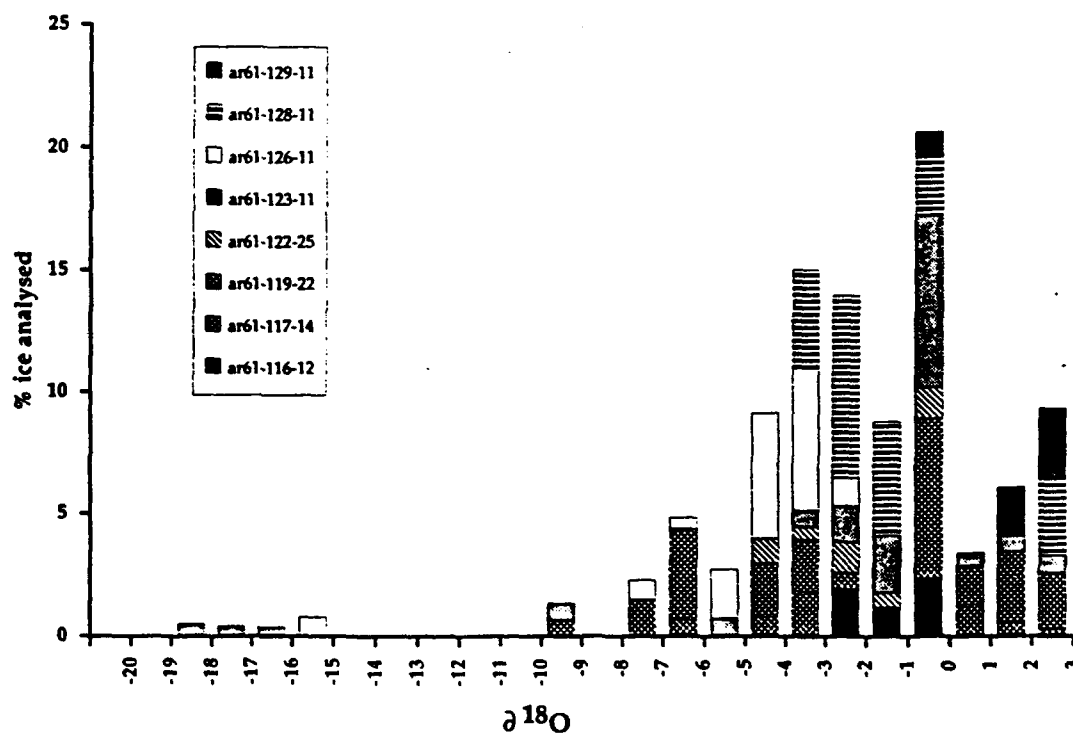
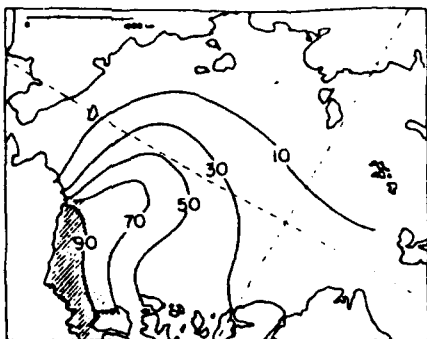


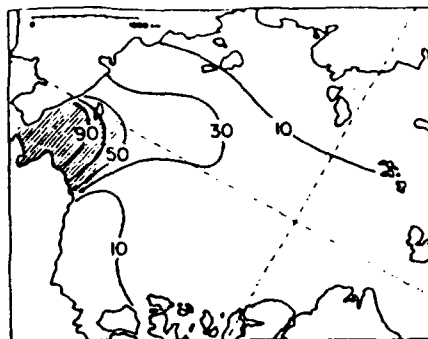
Figure 4. Eight cores obtained in the Barents Sea in May, 1989 generally had  $\delta^{18}\text{O}$  values less than 0‰ indicating that the ice was formed outside of the Barents Sea with influence from river water and/or precipitation. Less than 20% of the total 15 m of ice analyzed had  $\delta^{18}\text{O}$  values greater than 0‰ which is typical for ice growth in the Barents Sea. The very low values indicate refreezing of meltwater in the surface layers of some floes. Contaminants originally deposited on the ice surface may be redistributed by percolating meltwater.



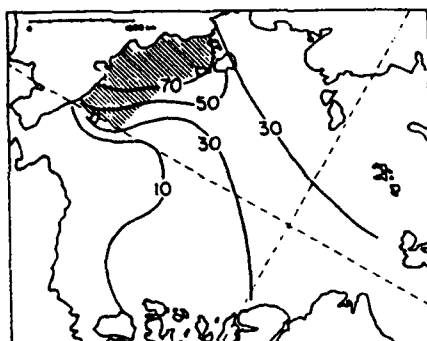
Beaufort Sea



Chukchi Sea



East Siberian Sea



Laptev Sea

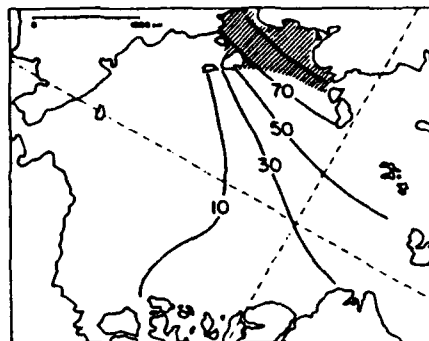


Figure 5. Contours of the asymptotic probability that ice formed in the shaded region moves into the Arctic Basin (Colony and Thorndike, 1985).

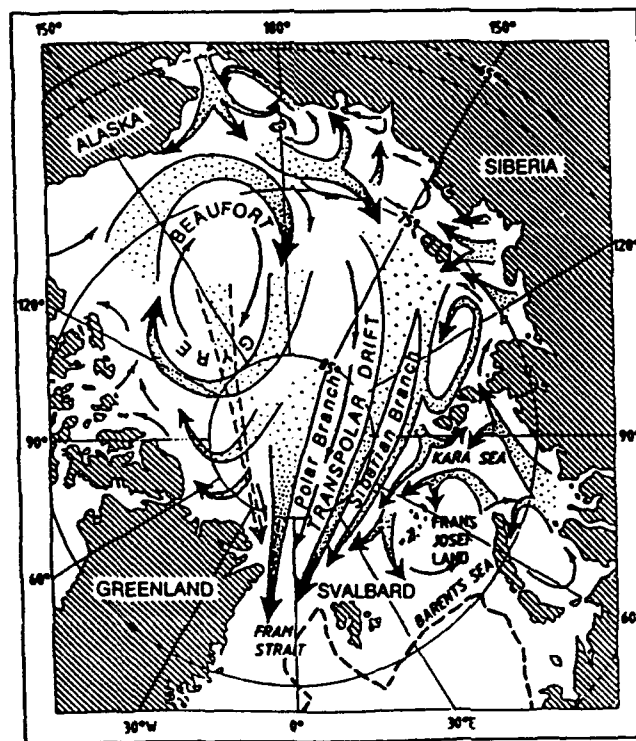


Figure 6. Mean sea ice drift patterns in the Arctic Ocean (Gordienko and Laktionov, 1969). Dashed line indicates the average maximum extent of sea ice.

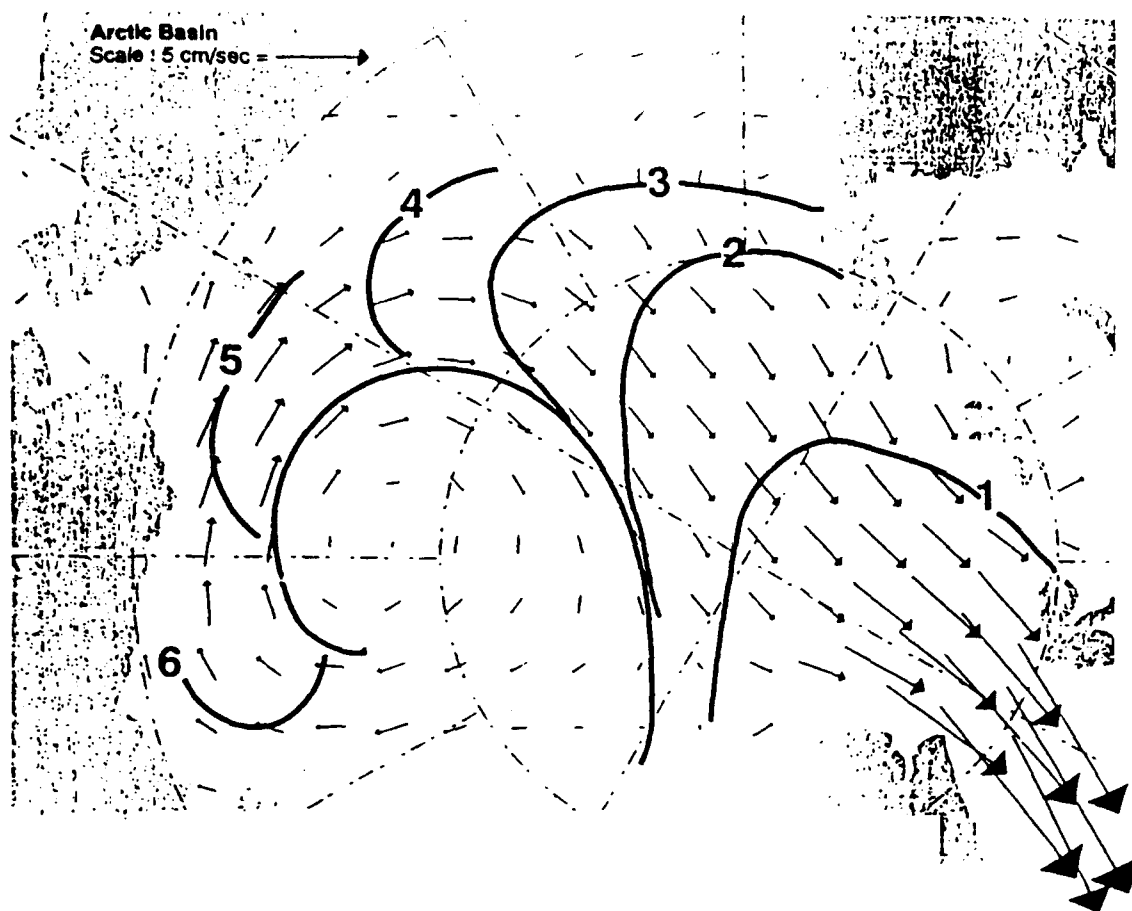


Figure 7. Annual mean ice motion in the Arctic Ocean during 1979-1990 based on drifting buoy data. The numbered lines show the expected time in years for the ice at that location to exit the Arctic Ocean through Fram Strait (Rigor, 1992).

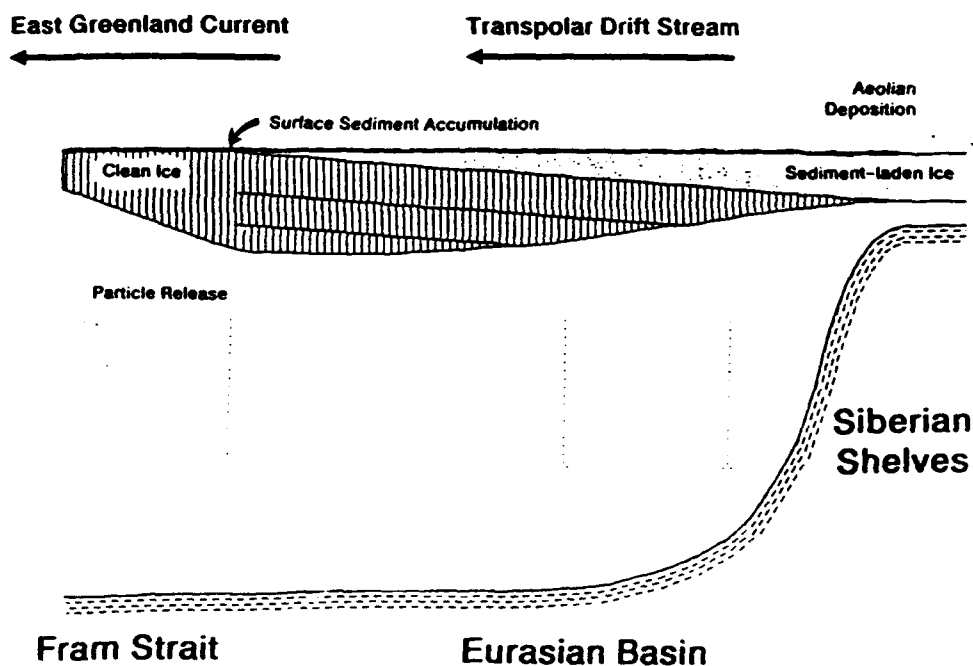
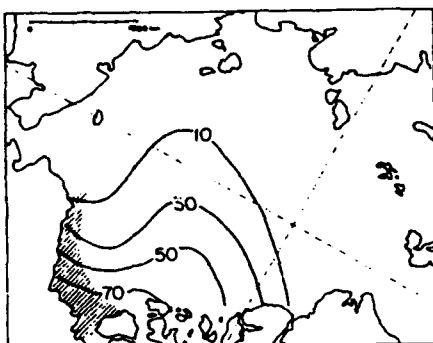
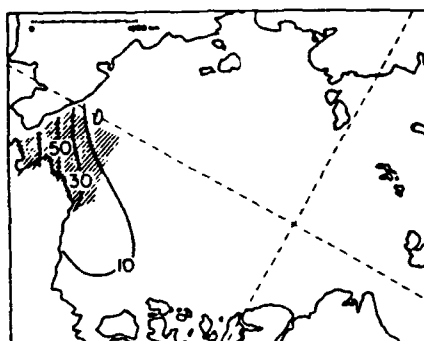


Figure 8. Schematic representation of ice growth, surface melting, and sediment accumulation during drift of a hypothetical ice floe from the Siberian shelf to the Fram Strait region (Pfirman, et al., 1990).

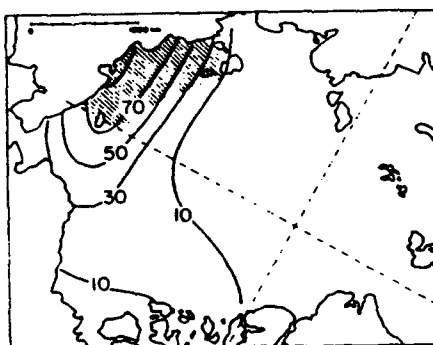
Beaufort Sea



Chukchi Sea



East Siberian Sea



Laptev Sea

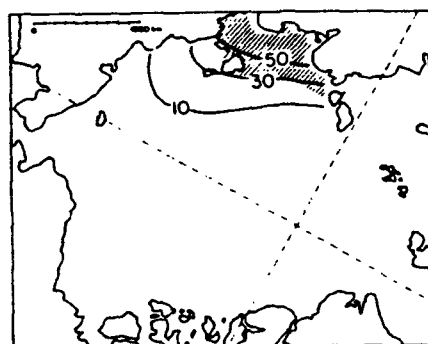


Figure 9. Contours of the asymptotic probability of ice moving into the shaded region and melting (Colony and Thorndike, 1985).

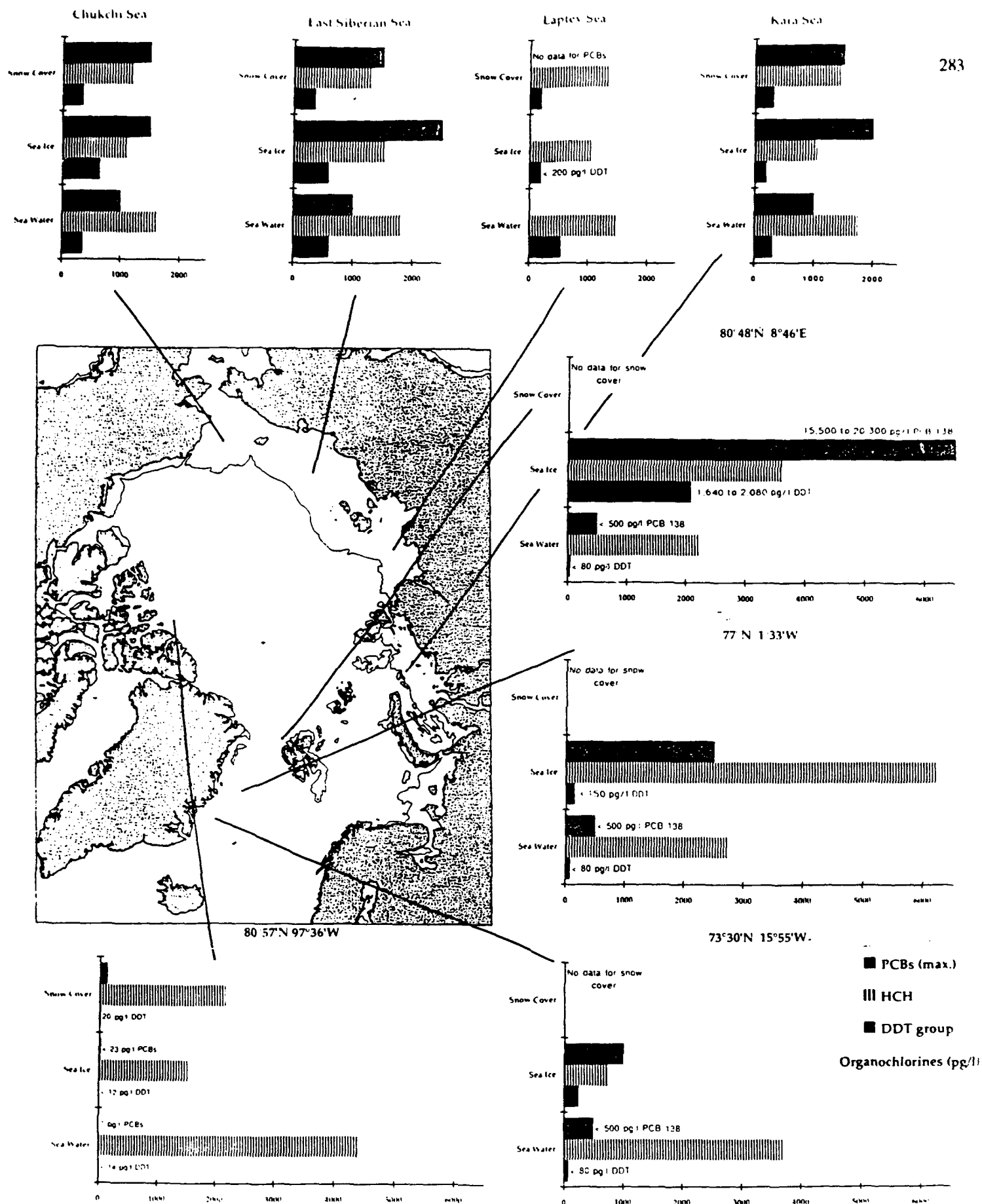


Figure 10. Organochlorine levels in Arctic snow, sea ice and surface water for the Chukchi, East Siberian, Laptev and Kara seas in 1990 (Melnikov, S.A. and S.V. Vlasov, 199?); north of Svalbard in 1979 and two stations on east Greenland shelf in 1985 (Gaul, in press); Canadian ice island in 1986 (Hargrave et al., 1988). The 100m bathymetric contour is shown to indicate the extreme shallowness of the eastern Kara, Laptev, Siberian and Chukchi seas.

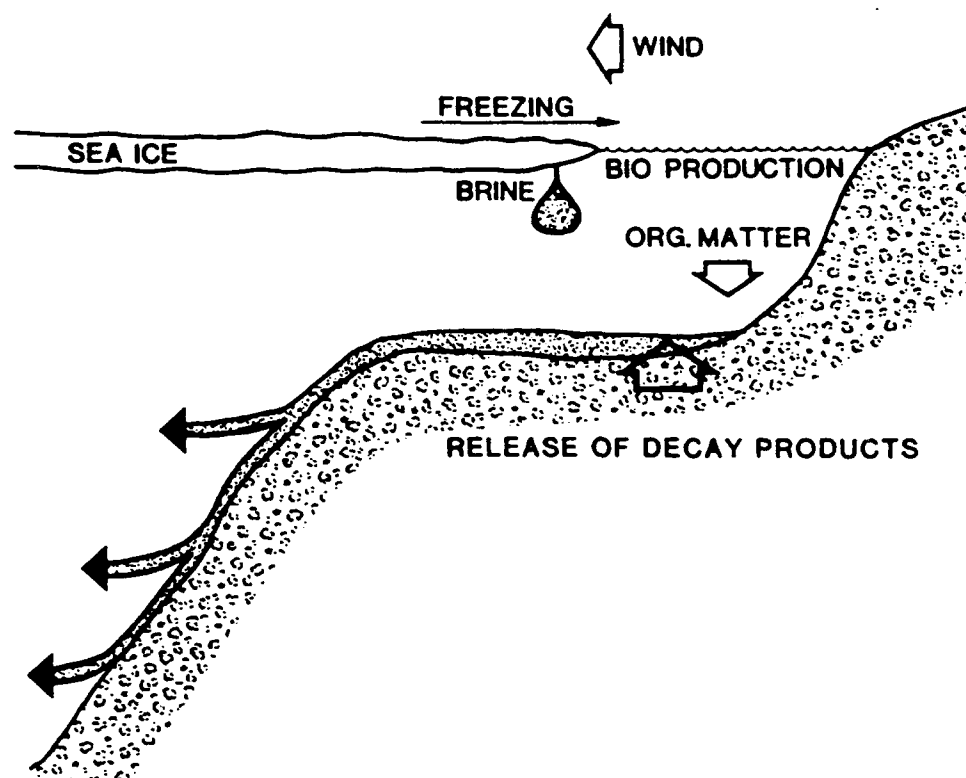


Figure 11. Brine rejection during sea ice formation leads to vertical convection over the shallow Arctic shelves (figure from Anderson and Dyrssen, 1989). The dense shelf brines formed in this way eventually run off the shelf, potentially carrying with them contaminated shelf sediments. Where convection extends from the sea surface to the sea floor and breaks down the stability of the water column, it also is easier for sea floor sediments to be resuspended higher in the water column.

## Large-scale circulation of the Arctic Ocean: implications for pollutant transport

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### Abstract

The key features of the large-scale circulation of the Arctic Ocean are reviewed based on distributions of hydrographic parameters and natural and anthropogenic trace substances. Mean residence times of the shelf waters in the Barents and Kara seas are discussed as well as potential pathways of pollutants released to the Siberian shelf seas from dumpsites or from river-runoff. Transit times needed for pollutants to cross the central Arctic basins are estimated from the distribution of the transient tracers tritium and its radioactive decay product  $^3\text{He}$ .

## 1 Introduction

Dumping of nuclear waste into the Siberian shelf seas, as well as discharge of radionuclides by Arctic rivers, poses a potentially serious pollution problem for the Arctic environment. Because observations are lacking, we cannot say with any certainty whether or not significant amounts of radioactivity have been released to Arctic shelf waters from dumpsites or by river-runoff. Assessment of the risk of spreading of radioisotopes after release from dumpsites or river deltas into the central Arctic basins and, via the East Greenland Current, into the Greenland/Norwegian seas requires

1. survey and monitoring of the shelf waters for radioactive contaminants, and
2. determination of pathways of Arctic shelf waters across the central basin and the time required for the transit to Fram Strait.

The best approach to solve these questions would be to trace a radionuclide which is released into the water column at the dumpsites. Potential nuclides to be measured are for example tritium,  $^{85}\text{Kr}$ ,  $^{90}\text{Sr}$ , or Cs isotopes ( $^{134}\text{Cs}$ ,  $^{137}\text{Cs}$ ). Existing radionuclide measurements show no clear evidence that radioisotopes have been released into Arctic shelf waters in significant amounts. Therefore, we cannot expect to be able to trace these radioisotopes directly.

In this case, we have to use our understanding of the Arctic circulation gained by a variety of oceanographic observations including the measurement of natural and anthropogenic trace substances to determine *potential* pathways of shelf waters and associated transfer times of these waters from the dumpsites to any other site within the Arctic Ocean or its adjacent seas.

Here, we provide an overview of the Arctic Ocean circulation and its implications for transport of pollutants released to the Siberian shelf seas. Additionally, we show how measurements of trace substances can be used to improve our capability to predict spreading patterns and transit times of potential pollutant releases.

## 2 Geographic setting

Bathymetry plays a key role in the dynamics of the Arctic Ocean. The Arctic Ocean is the smallest of the oceans. Several openings allow the Arctic Ocean to communicate with other water bodies (Fig. 1). Bering Strait is only 40 meters deep and 90 kilometers wide. Nares Strait is 250 meters deep and 15 kilometers wide, and the 130 meter deep Lancaster Sound connects the various waterways to the west of Ellesmere and Baffin Islands with Baffin Bay to the east. The largest gap in the land ring around the Arctic Ocean occurs between Greenland and Europe, where the Arctic Ocean abuts the Greenland and Norwegian seas. Svalbard breaks the open waters between the Arctic Ocean and the seas to the south into two distinctly different passageways. The 650 kilometer gap between Greenland and Spitsbergen is known as Fram Strait. With a sill at 2600 meters, Fram Strait provides the Arctic Ocean with its deepest level of direct contact with other oceans. The bathymetry of the 675 kilometer gap between Svalbard and Scandinavia is a continental shelf which limits the exchange through this passageway to the upper 200 meters.

The wide continental shelf of Eurasia is another unique feature of the Arctic Ocean. At its widest point in the area between the island of Novaya Zemlya and Svalbard, the continental shelf width exceeds 900 kilometers, the widest shelf in the world. The waters above the Eurasian continental shelf are divided into five shelf seas by various islands and continental protrusions: Barents Sea, Kara Sea, Laptev Sea, East Siberian Sea and Chukchi Sea.

The St. Anna Trough and the Voronin Trough are steep deep-sea canyons cut into the continental shelf between the islands of Franz Joseph Land and Svernaya Zemlya. These canyons have been the subject of oceanographic interest since the turn of the century as a possible site for communication between the deep Arctic Ocean and the shallower shelf seas. For example, dense waters which form in the shelf seas might flow out into the central Arctic through the canyons.

The bathymetry of the deep Arctic Ocean itself influences the direction and magnitude of flow. The most important in oceanographic terms is the Lomonosov Ridge, which divides the Arctic Ocean into two major sectors, the Eurasian Basin and the Canadian Basin. The sill depth along the ridge remains the subject of conjecture, but seems to be somewhere between 1400 and 2000 meters.



Two deeper, more fractured oceanic ridge systems, one in the Eurasian Basin and one in the Canadian Basin, again divide each basin into distinct smaller basins. The Gakkel ridge system parallels to the Lomonosov Ridge at a depth of about 2800 meters, splitting the Eurasian Basin into two basins of approximately equal area, the Amundsen Basin toward the ocean's center and the Nansen Basin toward the continental fringe. Within the Canadian Basin, the ridge systems of the Mendeleyev Ridge and the Alpha Ridge split that basin into the narrow Makarov Basin toward the Lomonosov Ridge and the larger Canada Basin extending toward Alaska.

The deep basins of the Arctic Ocean have average depths of about 4000 meters.

### 3 Physical conditions

The seas of the Arctic region are linked by common threads in their geography and climate that distinguish them from the other parts of the ocean:

1. Arctic seas are 'mediterranean' in that they are almost entirely surrounded by land. Because of this, exchanges with other oceans are limited and occur mostly through narrow and relatively shallow channels. The Arctic Ocean is a mediterranean sea of the Atlantic Ocean, a salty, warm ocean. The Atlantic is well oxygenated, has low nutrient concentrations and its waters are renewed on a relatively short time scale (compared to the other major ocean basins).
2. About 30 % of the total surface area of the Arctic Ocean is continental shelf. The continental shelves are the site of river input, of seasonal fluctuations in sea-ice cover, and of interactions with the sea bed, all key elements of the dynamics of the Arctic Ocean.
3. The presence of the permanent sea-ice cover over the central Arctic places an insulating lid over much of the interior, limiting exchange with the atmosphere to relatively infrequent leads.
4. The waters of the Arctic region are mostly quite cold. This means that variations in density are controlled primarily by salinity — as opposed to the subtropical gyres where density largely depends upon

temperature. A direct implication is that changes to the salt budget or salt stratification can have profound effects on the region.

The cold Arctic waters are also subject to increased effectiveness of processes which depend upon the intricacies of the equation of state at low temperatures.

5. At very high latitudes the variation of the Coriolis parameter with latitude is small, and so we might expect unusually strong coupling of the circulation to bathymetry.

## 4 Hydrographic structure

The main features of the hydrographic structure are discussed on the basis of the distributions of potential temperature, salinity, and density referred to zero and 3000 decibars on a long section extending north from the Faeroe-Shetland Channel, through the Norwegian and Greenland seas, through Fram Strait, and across the Eurasian and Canadian basins to the Alaskan continental shelf (Fig. 2). The data used in this figure are few and mostly old, but the basic features are the same as seen in the few high resolution data sets that have only recently become available. We will concentrate our discussion on the Arctic Ocean parts of the section.

Nansen (1906) classified three water masses found everywhere in the polar basin: upper water, Atlantic layer, and deep water.

### 4.1 Upper waters

The upper waters can be divided into the Polar mixed layer (30 to 50 m depth) and the halocline (about 30 to 50 m to about 200 m depth).

In the Polar Mixed Layer the temperature is generally close to the freezing point. This layer represents local mixing of waters below the sea-ice cover. River-runoff contributes freshwater to the upper part of the water column. The total mean annual river transport of 0.11 Sv spilling into the Arctic Ocean seems small, but the fresh, low density waters have a profound impact on the small, cold and relatively isolated Arctic Ocean. River waters float over the saline oceanic waters, increasing the density gradient within the upper ocean. The cumulative effect of other fresh water inputs, such as ice

melt, excess precipitation and the relative freshness of the Bering Sea inflow, which is in part due to outflow from the Yukon River, produce some of the freshest surface waters in the world oceans. (Fig. 3).

When sea water freezes, most of its salt is released. The released brine is heavier than the waters directly below the ice and so the brine sinks. While descending, brine mixes with fresher near surface waters. Tritium studies by Östlund (1982) indicated that the influence of this melting-freezing cycle extends 10-60 meters into the water column in the Nansen Basin and can reach 150 meters in the Canadian Basin. Wind and stirring by ice keels help to mix the released brine with surrounding waters.

Below the Polar mixed layer is the halocline. Temperatures within the lower part of the halocline are warmer than the near freezing temperatures of the upper halocline and the surface mixed layer but they do not exceed 0°C.

The halocline is permanent over the deep basins — it is never penetrated as far as we can tell. Therefore, it is an effective shield against heat flux from the relatively warm Atlantic layer below into the surface mixed layer. Waters in the halocline form on marginal shelf seas and are advected the central Arctic basins. Halocline waters are formed during the winter freeze. Highest rates of brine production occur in areas of divergence, where sea-ice constantly forms but is pushed away allowing more fresh sea-ice to form and release salt. The resulting high salinity, low temperature water masses flow off the shallow shelf. In this way, the waters of the halocline are renewed from the periphery of the Arctic Ocean.

Nutrient signals in the halocline support the conjecture of shelf sea origins. High silicate values within nutrient maxima of the upper halocline in Canadian Basin stations partly point to a Pacific Ocean source. However, inflowing Bering Sea Water nutrient values are not high enough themselves to be the sole source. Jones and Anderson (1986) suggested that regenerated biogenic material on the shelves added to Bering Sea Water flowing off the shelves into the Arctic Ocean accounts for this high nutrient feature being seen in locations far from the Bering Strait.

Jones and Anderson's physical and chemical studies of the more salty water of the lower halocline typically found in the Eurasian Arctic point to the Barents and Kara shelves as source sites. These shelves provide nutrient regeneration from decayed biomass but with a shorter residence time and consequently lower nutrient concentrations. Also, mixing with water from

the saline, less nutrient-rich Atlantic Ocean accounts for the higher salinity.

## 4.2 Atlantic layer

Below the upper layer, there is a warm and salty layer which is known as the Atlantic Layer because of its origin in the Atlantic Ocean. The Atlantic layer is identified most easily by a mid-depth temperature maximum, ranging from about 0.5 °C to 4 °C. This layer originates in the Faeroe-Shetland inflow, but as it circulates northward it cools, and sinks beneath the much fresher polar waters in Fram Strait. Traditionally, the boundaries of the Atlantic Layer have been defined by the 0 °C isotherm. Within the Arctic Ocean, the Atlantic layer typically is found between about 200 and 800 m depth.

## 4.3 Deep waters

Below the Atlantic Layer, the deep waters of the Arctic Ocean are marked by high salinity and cold temperatures. The deep waters make up 60 % of the volume in the Arctic Ocean. The distribution of deep waters are influenced by the Lomonosov Ridge. Colder, fresher, higher density deep waters are kept from major influence on the Canadian Basin. The deep waters in the Canadian Basin represent one extreme in the Arctic seas. The other deep extreme is the Greenland Sea, which, at least occasionally, experiences ventilation from the surface to the deep waters. The Greenland Sea on the whole is very cold, fresh, and dense. It has the youngest deep water in these seas (e.g. Heinze et al., 1990; Schlosser et al., 1991), while the Canada Basin has the oldest deep water (e.g. Östlund et al., 1987; the Eurasian Basin deep water has mean residence times between these two extremes; Schlosser et al., 1990).

The Arctic Ocean is, on the whole, quite a bit warmer and saltier than the Greenland Sea. Deep water salinity values in the Arctic Ocean are too high to be explained solely by lateral advection from the Greenland and the Norwegian seas. Greenland Sea Deep Water (GSDW) has an average salinity of 34.89 and Norwegian Sea Deep Water (NSDW) an average salinity of 34.91. Yet, all salinities measured in the central Arctic Ocean are in excess of about 34.93 below 1500 meters. An additional source of salt from within the Arctic Ocean is necessary to explain these values.

Salt could be added to the deep basins from the shelves. Very dense, saline waters have been found on the shelves, but no one has observed brine-enriched waters sinking to depth at the periphery. Theoretically, the observed shelf waters are not heavy enough to sink deep into the central basins of the Arctic Ocean without mixing or simply hugging the boundary. High  $\delta^{18}\text{O}$  values measured in the deep waters may rule out shelf sources influenced by extensive river-runoff, and most shelf seas contain extensive amounts of river runoff (e.g. Grabitz et al., 1993).

Although there are many questions concerning the formation of the deep water in the Arctic Ocean, contributions from the Arctic's peripheral shelves seems necessary to explain the high salt values in the deep basins.

The freshness of the Eurasian Basin in comparison to the Canadian Basin may be explained by the proximity of the Eurasian sector to cold, fresher GSDW and NSDW. The Lomonosov Ridge excludes any GSDW or NSDW from directly entering the Canadian Basin. Recent  $^{14}\text{C}$  measurements show the apparent age of Canada Basin ( $\approx 700$  years; Östlund et al., 1987) waters to be considerably older than Eurasian Basin waters ( $\approx 250$  years; Schlosser et al., 1990).

#### 4.4 Link to the adjacent seas

The Greenland and Iceland gyres take warm salty upper layer waters from the North Atlantic past sills, underneath cold air, and transform this surface water into various colder and denser water masses. These dense waters recirculate in the Arctic region and also overflow into the North Atlantic over the Greenland-Scotland sills.

The continental shelves are also exposed to the atmosphere for much of the year, and the shallow water column can manufacture dense water quickly. These may cascade down the slopes into the deep basins.

Exchanges between the deep basins are limited or controlled by bathymetry. The most isolated basin is the Canada Basin, far removed from the open-ocean ventilation in the Greenland and Iceland seas.

## 5 Arctic Ocean circulation

### 5.1 Upper waters

The circulation of the surface waters of the Arctic Ocean is increasingly well understood based on studies of sea-ice drift. Direct measurements of water velocities within the Polar Mixed Layer indicate that the layer moves on average with the overlying sea-ice. Also, computations of dynamic height near the surface relative to deeper water show the same large-scale flow features as the observed ice drift (Coachman and Barnes, 1963).

The prominent long term features of the Arctic ice pack drift are the anti-cyclonic Beaufort Gyre occupying most of the Canadian Basin and the Transpolar Drift flowing from the pole toward Fram Strait (Fig. 4).

Circulation patterns in the halocline waters in the central Arctic Ocean have to date been studied only with core methods, by tracing property maxima or minima. Flow appears to be around to the right on the shelves owing to the Coriolis force. Once off the shelves, flow must penetrate into the central Arctic Ocean to maintain the cold halocline structure. But we do not yet have a map of the circulation in this key layer.

### 5.2 Atlantic layer

The Atlantic layer enters the Arctic Ocean with temperatures greater than 3°C and salinities above 35. The temperature maximum is maintained as the water travels around the Arctic Ocean. Coachman and Barnes (1963) used the core method to trace the flow pattern, which was found to be generally cyclonic (Fig. 5). Further investigations suggest that much of the subsurface flow occurs in boundary currents and that, in general, the flow in all subsurface layers, i.e., beneath the halocline, follows this pattern. We expect that the Atlantic layer is profoundly influenced by processes in the Barents and Kara seas, and perhaps even the rest of the Siberian shelf seas.

The water of Atlantic origin exits the Arctic Ocean at intermediate depths in the East Greenland Current.

### 5.3 Deep waters

Due to the paucity of high quality deep data, the sense of circulation in the deep layers is not well observed. Aagaard (1989) deduced from mooring data a cyclonic sense of flow around the Eurasian basin along the periphery at an average speed of 4 cm/s, with its speed increasing with depth to the deepest measurement at 1000 meters. In the Canada basin, Beaufort Sea moorings also indicated a general cyclonic flow trapped against the slope, although the flow did not extend as deep as did the Eurasian flow. Aagaard noted that the measurements supported the assessment of Manley and Hunkins (1985), much of the flow in the Canada Basin below the mixed layer is dominated by the mesoscale eddy field.

### 5.4 Transport rates

Transports through Fram Strait are probably something like 2 to 5 Sv on each side, with northward flowing waters on the east and southward flowing waters on the west. The picture is greatly complicated by recirculation in the passage, natural variability, and measurement problems. Transports through other passages are smaller: about 1.5 Sv into the Arctic through Bering Strait, 2.1 Sv out of the Arctic through the Canadian Arctic Archipelago, and 0.6 Sv into the Arctic over the Scandinavian continental shelf.

### 5.5 Overall circulation scheme

The overall circulation scheme within the Arctic Ocean is characterized by the following key features (Fig. 6):

1. In the deep basin regions which are largely ice free such as the Greenland and Iceland seas, a balance between the inflowing waters, sea-ice cover, and air/sea exchange prevails which permits convection to intermediate and in some cases abyssal depths.
2. Most of the peripheral shelf seas have great seasonal fluctuations in their sea-ice cover. The brine released by sea-ice formation can accumulate on the shelf and mix with waters around the periphery to produce dense water.

3. There are deep basin regions covered by sea-ice. We know these regions must be relatively passive, because they are largely insulated.

## 6 Implications for pollutant transport

Prediction of pollutant transport and transit times from a potential release site in one of the shelf seas to other parts of the Arctic requires knowledge of:

1. The mean residence times of the shelf waters.
2. The potential pathways of the shelf waters off the shelf seas and across the central basins.
3. The time required for transit from the shelf seas to any other point in the Arctic Ocean or its adjacent seas.

Due to the sparse observations available from the Arctic Ocean, we do not yet understand the coupling of the shelf regions to the interior, and especially how, or perhaps even if, the deep waters are coupled to the shelf outflows.

It is therefore very difficult to judge in advance the most likely dispersal pathways and transit times for effluents from any of the shelf seas to other parts of the Arctic Ocean. This is a major gap in our capability to predict the fate of pollutants released to the Arctic Ocean that will take careful observational, modeling, and theoretical studies to sort out.

Although we cannot precisely predict the fate of pollutants released to the Arctic shelf seas, we can use the distribution of several natural and anthropogenic trace substances in the Arctic Ocean to obtain information on likely pathways and transit times.

### 6.1 Mean residence of the shelf waters in the Barents and Kara seas

Several estimates of the mean residence time of freshwater in the Barents and Kara seas based on different methods are available. Recently, Schlosser et al. (1993) derived a mean residence time of the Arctic river-runoff on the shelves of the Kara and Barents seas of about  $3.5 \pm 2$  years. This value was



based on tracer observations and is in good agreement with values derived from salinity balances (about 3 years: Aagaard and Coachman, 1975; about 2.5 years: Hanzlick and Aagaard, 1980). This means that pollutants injected into the shelf waters of the Kara or Barents seas could leave these regions within approximately 3 years.

## 6.2 Potential pathways of pollutants

It seems to be a safe assumption that most of the shelf waters exit the Arctic Ocean in the upper water layers, i.e. above the Atlantic layer. The upper waters are characterized by relatively low salinities indicating a relatively large freshwater component. The freshwater component consists of sea-ice meltwater and river-runoff. Most of the Arctic river-runoff is added to the shelf seas east of the Barents Sea (Kara, Laptev and East Siberian seas). River-runoff is characterized by low  $\delta^{18}\text{O}$  values (about  $-21\text{‰}$ ; e.g. Östlund and Hut, 1984). This isotopic signature can be used to trace the pathway of shelf waters containing a significant river-runoff component across the central Arctic Ocean through Fram Strait into the East Greenland Current (e.g. Östlund and Hut, 1984; Schlosser et al., 1993; Grabitz et al., 1993). Based on salinity and  $^{18}\text{O}$  balances we can estimate the fraction of river-runoff and sea-ice meltwater contained in the Arctic Ocean surface and halocline waters. The distribution of the river-runoff signal can then be used to trace the river-runoff signal across the central Arctic basins. This signal is also a good indicator for any pollutant released to shelf waters containing a significant amount of river-runoff.

Based on the presently available data from the central Eurasian Basin, we can conclude that the river-runoff signal is confined to the northern part of the Nansen Basin (Fig. 8; for geographical position of the stations, see Fig. 7), i.e. it seems to follow the Transpolar Drift. Therefore, pollutants released to the shelf waters east of the Barents Sea seem to cross the central Arctic basins north of the central Nansen Basin. Due to the lack of data from the Canadian Basin, we do not know any details of the circulation of river-runoff in the upper layer of this part of the Arctic Ocean.

Our present data set allows us to define a sharp southern boundary of what could be a plume of river water crossing the central Arctic Ocean. As our sections do not reach far enough into the Canadian Basin, we cannot yet say whether there is another boundary in the Canadian Basin (most likely

coinciding with the boundary between the Transpolar Drift and the Beaufort Gyre) which would confine the river water in a well-defined 'stream' or if the river-runoff is more dispersed in this part of Arctic Ocean.

Shelf waters without a significant river-runoff fraction (typically Barents Sea waters) are observed in the southern Nansen Basin. They are separated from the waters carrying the river-runoff signature by a relatively sharp front at about 83.5°N in the center of the Nansen Basin (Schlosser et al., 1993).

Once a representative  $^{18}\text{O}$  data set has been established for the entire Arctic Ocean, we can use  $^{18}\text{O}$ /salinity balances to describe the spreading of the river plumes across the central Arctic basins (for the Canadian basin, an additional parameter, most likely silicate, is required for separation of the river-runoff component from other freshwater sources; e.g. Grabitz et al. 1993)

### 6.3 Mean residence times of the Arctic waters

The mean residence times of the Arctic waters span a wide range: from less than a year to several hundred years. Naturally, near-surface waters are renewed much more rapidly than deep waters. As most of the dissolved pollutants are transported in the upper water column, the mean residence times of the surface and halocline waters are most relevant for estimates of transit times of pollutants.

The average mean residence time of freshwater in the upper layers of the Arctic Ocean is about 10 years (e.g. Aagaard and Coachman, 1975; Östlund and Hut, 1984). However, recent measurements of transient tracers in the surface waters and the Arctic halocline (e.g. Wallace et al., 1987, 1992; Schlosser et al., 1990, 1993) suggest that the surface waters are renewed much faster (about 2 to 6 years) than the halocline waters (up to 15 years; Fig. 9). This means that dissolved pollutants entering the upper layers of the Arctic Ocean from the shelf seas could exit through Fram Strait in about 5 years, and pollutants released during a single event should be flushed out of the surface waters on time scales of a few decades.

Intermediate and deep water renewal times are much larger. We can use bomb tritium as a proxy to estimate time scales for water mass transport from the shelves to these water layers. Bomb tritium enters the central Arctic Ocean mainly from the shelf seas. Shelf waters flowing down the continental slope are concentrated in boundary currents from which they

mix into the interior of the basin (e.g. Smethie et al., 1988; Anderson et al., 1989). This results in tritium concentrations in the deep Nansen Basin that are higher at the boundaries than in the interior (Fig. 10).  $^{39}\text{Ar}$  data (Buehler, 1993) and model-based evaluation of the transient tracer fields (e.g. Heinze et al., 1991; Bönisch, 1991) show that the deep and bottom waters away from the boundaries have mean apparent ages of about 50 (Eurasian Basin Deep Water) to 300 (Eurasian Basin Bottom Water) years. The mean apparent ages of the deep waters in the Makarov and Canada basins are about 350 to 450 years (Bühler, 1992; Kromer and Schlosser, unpublished  $^{14}\text{C}$  data) and 700 years (Östlund et al., 1987), respectively.

## 7 Conclusions

From available measurements we can conclude that if radionuclides are released in the Barents and Kara seas in dissolved form, they will to a large extent be transported out of the Arctic ocean in the shallow water column on a time scale of several years to 1 or two decades. The fraction that will penetrate the deeper waters will be smaller than the surface inventory and will reside in the Arctic Ocean for many decades to several centuries.

## Acknowledgements

This work was supported by the Office of Naval Research and the National Science Foundation.

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Fig. 1: Bathymetry of the Arctic Mediterranean (depths in meters). The long line extending from the southern Norwegian Sea to the southern Canadian Basin locates the section shown in Fig. 2. Reprinted from *Journal of Geophysical Research*, Volume 90, K. Aagaard, J.H. Swift, and E.C. Carmack, Thermohaline Circulation in the Arctic Mediterranean Seas, pages 4833- 4846, copyright 1985, American Geophysical Union.

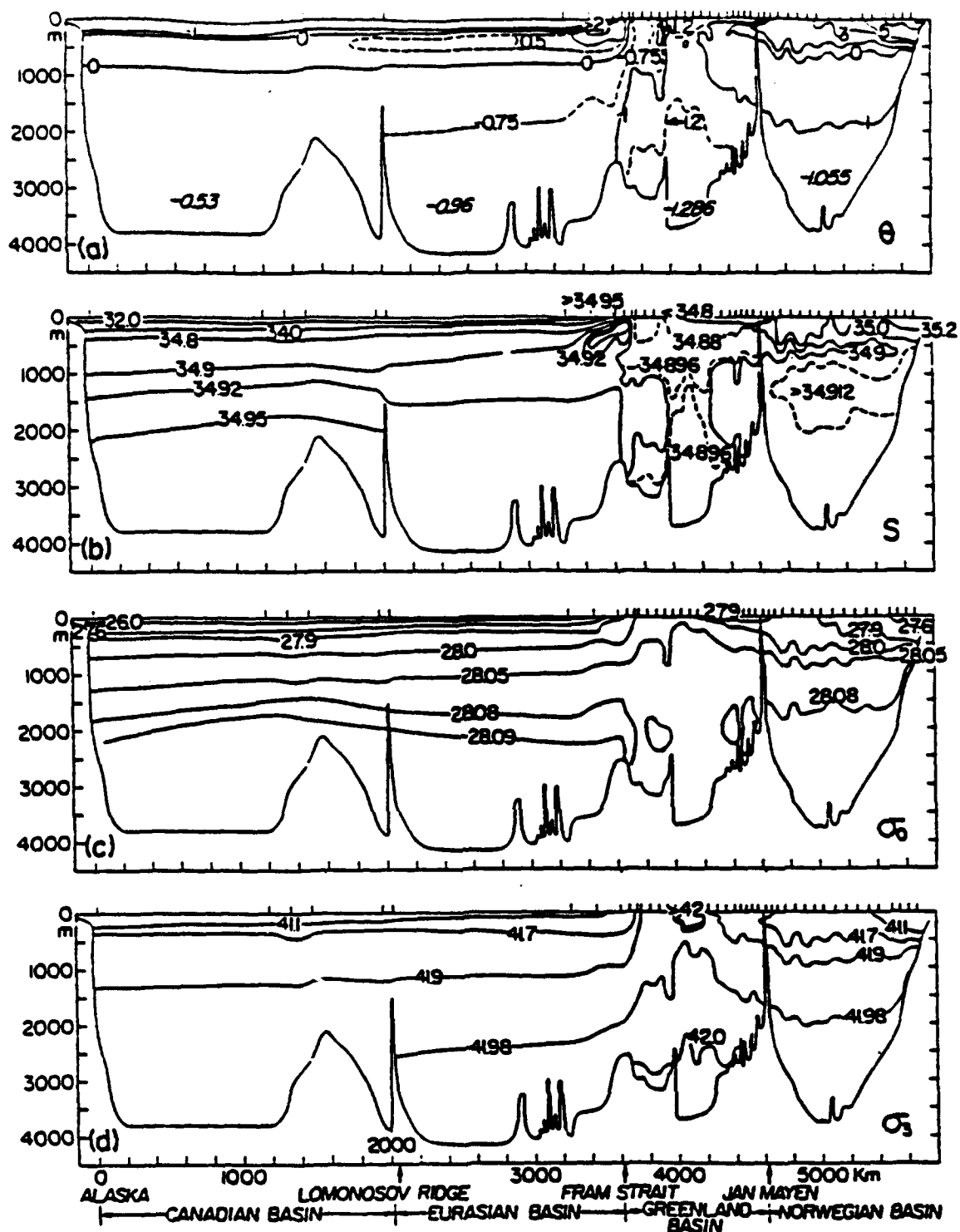


Fig. 2. Distributions of potential temperature, salinity, and potential density anomaly referenced to 0 and 3000 decibars along the section shown in Fig. 1. Reprinted from *Journal of Geophysical Research*, Volume 90, K. Aagaard, J.H. Swift, and E.C. Carmack, Thermohaline Circulation in the Arctic Mediterranean Seas. pages 4833- 4846, copyright 1985, American Geophysical Union.



Fig. 3. Salinity (psu) at the sea surface. Reprinted from Deep-Sea Research, 26, J.L. Reid, On the contribution of the Mediterranean Sea outflow to the Norwegian-Greenland Sea, pages 1199-1223, copyright 1979, Pergamon Press Ltd.



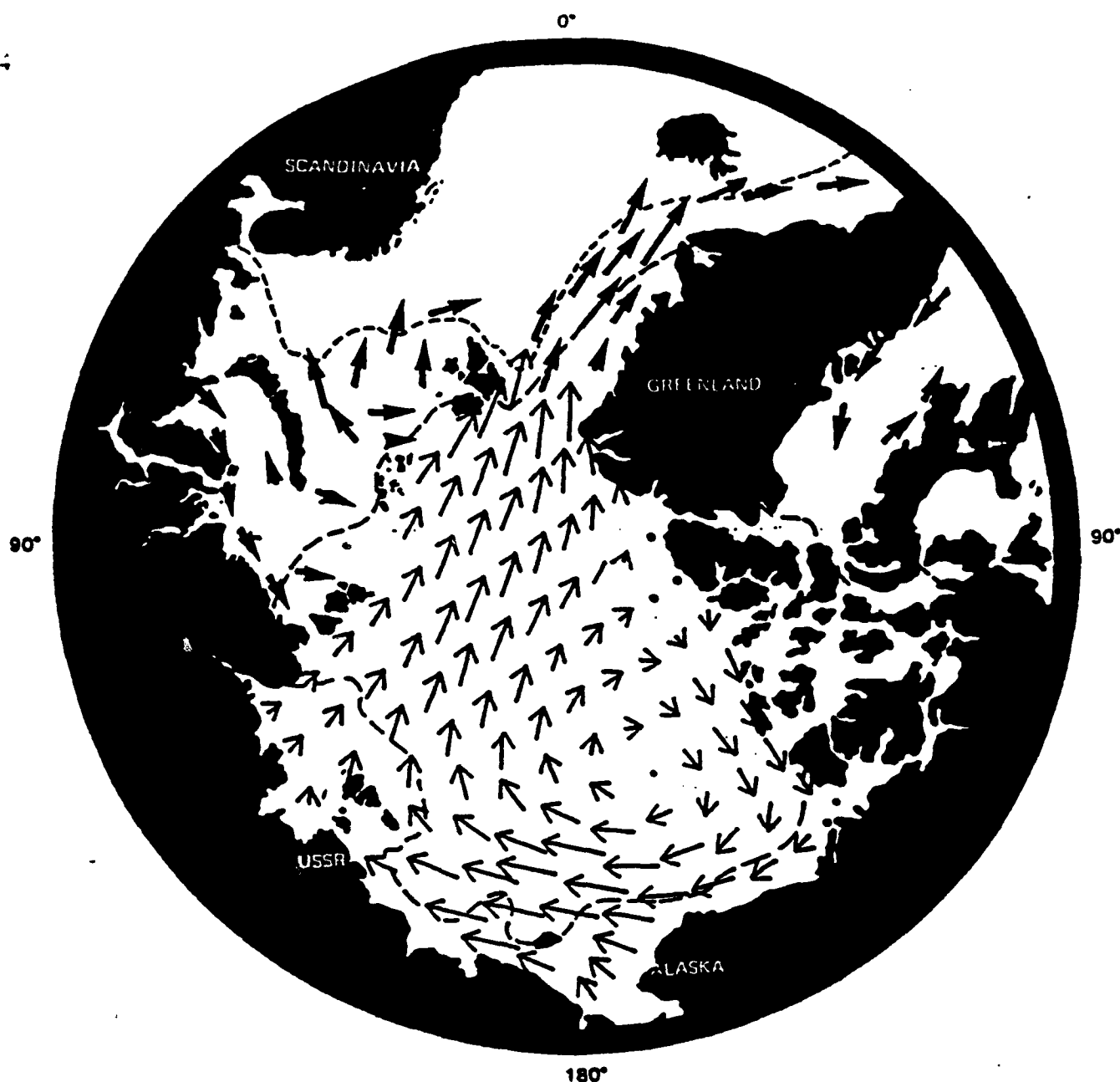


Fig. 4. Maximum (dotted lines) and minimum (dashed lines) ice extent in the Arctic Ocean; and mean sea ice drift (arrows), derived mainly from automatic data buoy tracks (from Polar Science Center, University of Washington). Reprinted from *Polar Marine Diatoms*, L.K. Medlin and J. Priddle, eds., E.C. Carmack and J.H. Swift, *Some Aspects of the Large-Scale Physical Oceanography of the Arctic Ocean Influencing Biological Distributions*, pages 35-46, copyright 1990, British Antarctic Survey.

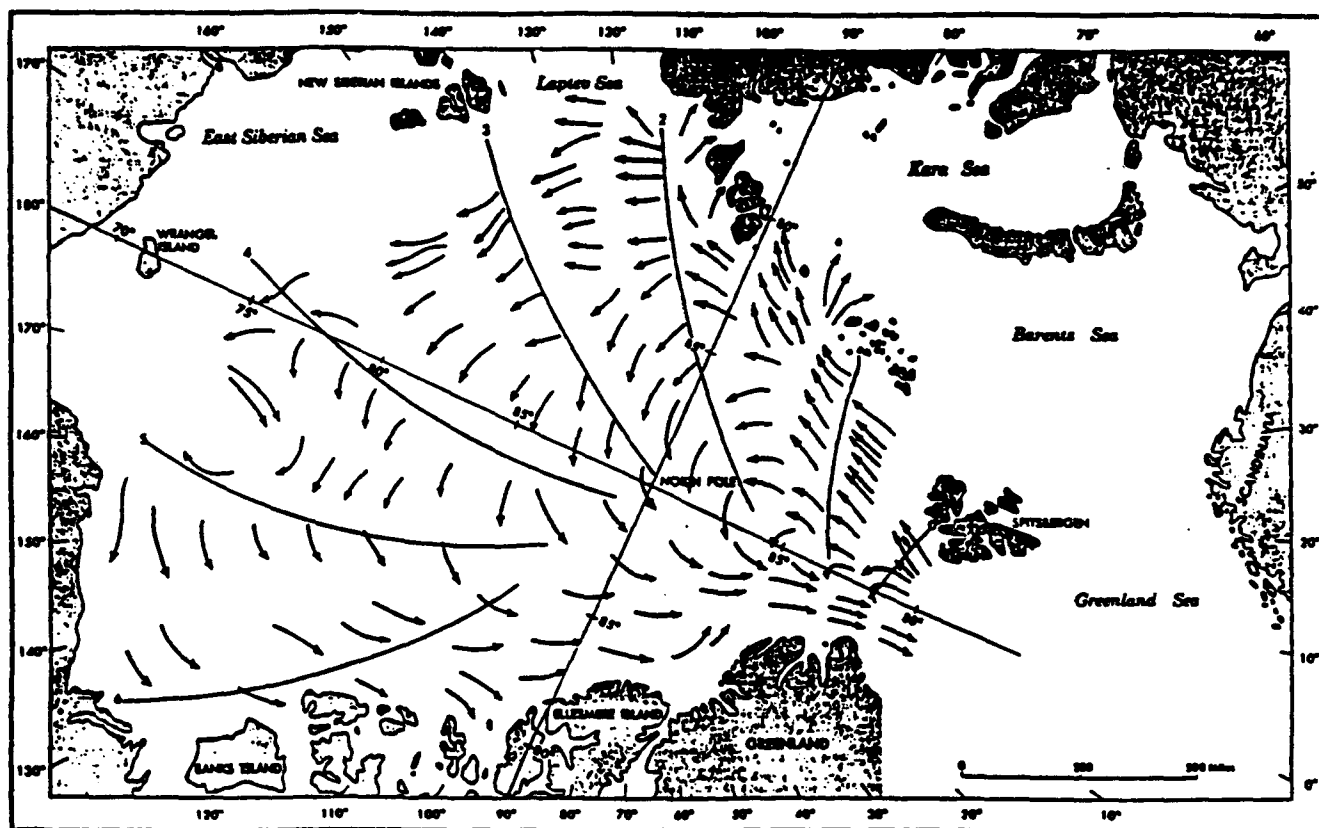


Fig. 5. Circulation of the Atlantic layer. Reprinted from *Arctic*, Volume 16, L.K. Coachman and C.A. Barnes, *The movement of Atlantic water in the Arctic Ocean*, pages 8-16, copyright 1963, Arctic Society of North America.

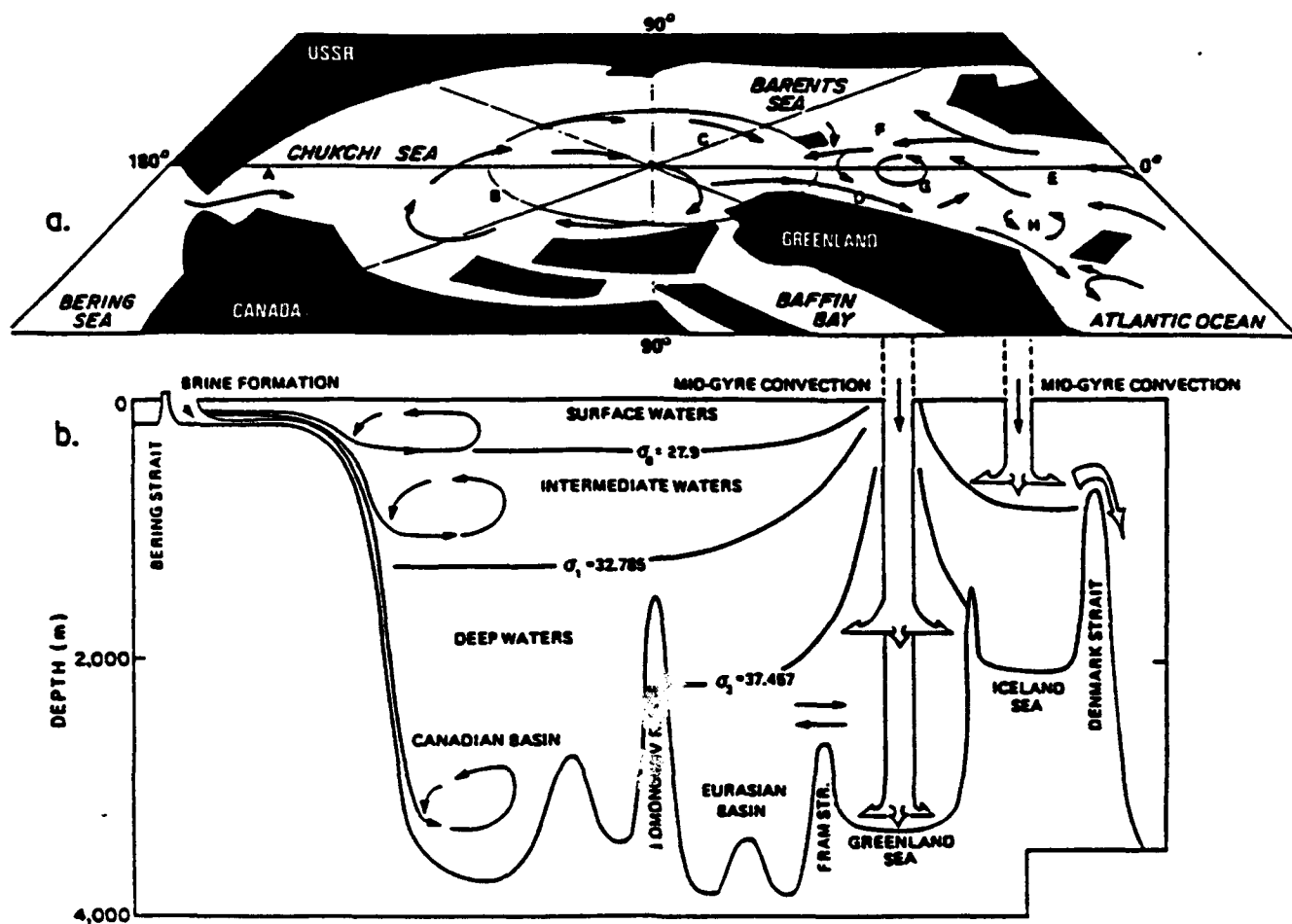


Fig. 6. Schematic picture of the circulation in the Arctic Ocean. Reprinted from *Journal of Geophysical Research*, Volume 90, K. Aagaard, J.H. Swift, and E.C. Carmack, Thermohaline Circulation in the Arctic Mediterranean Seas, pages 4833- 4846, copyright 1985, American Geophysical Union.

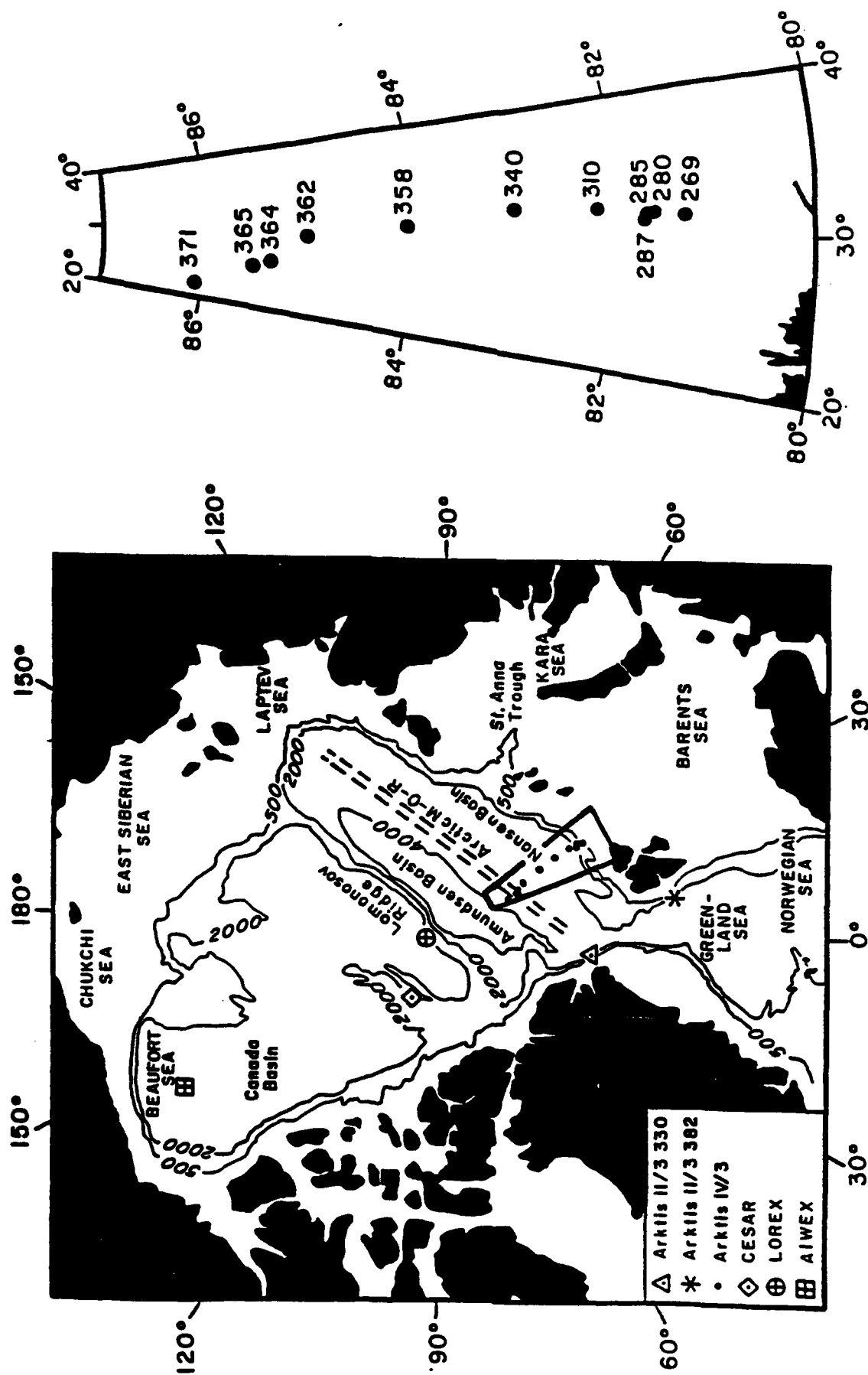


Fig. 7: Geographic positions of the ARK IV/3 stations used in Figs. 8 and 9.

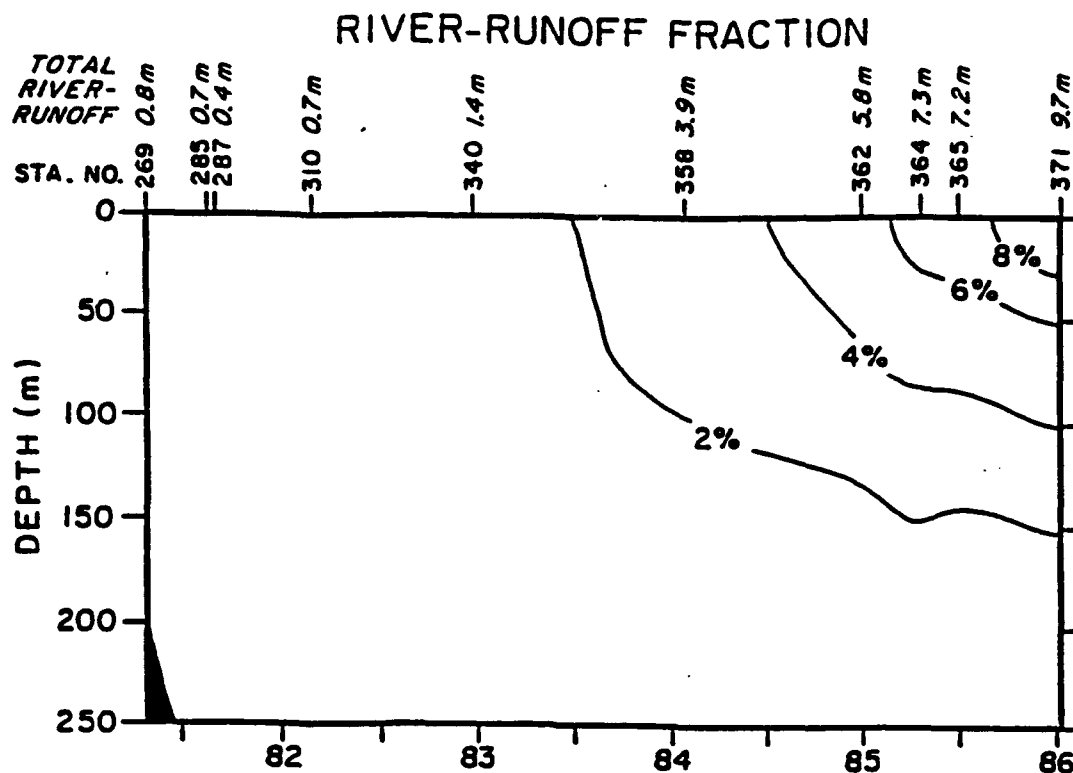


Fig. 8: River-runoff fraction contained in the upper 250 meters of the water column along a section extending from the Barents Shelf (sta. 269) to the Gakkel Ridge (sta. 371; for geographical position of the stations, see Fig. 7). From Schlosser et al., 1993.

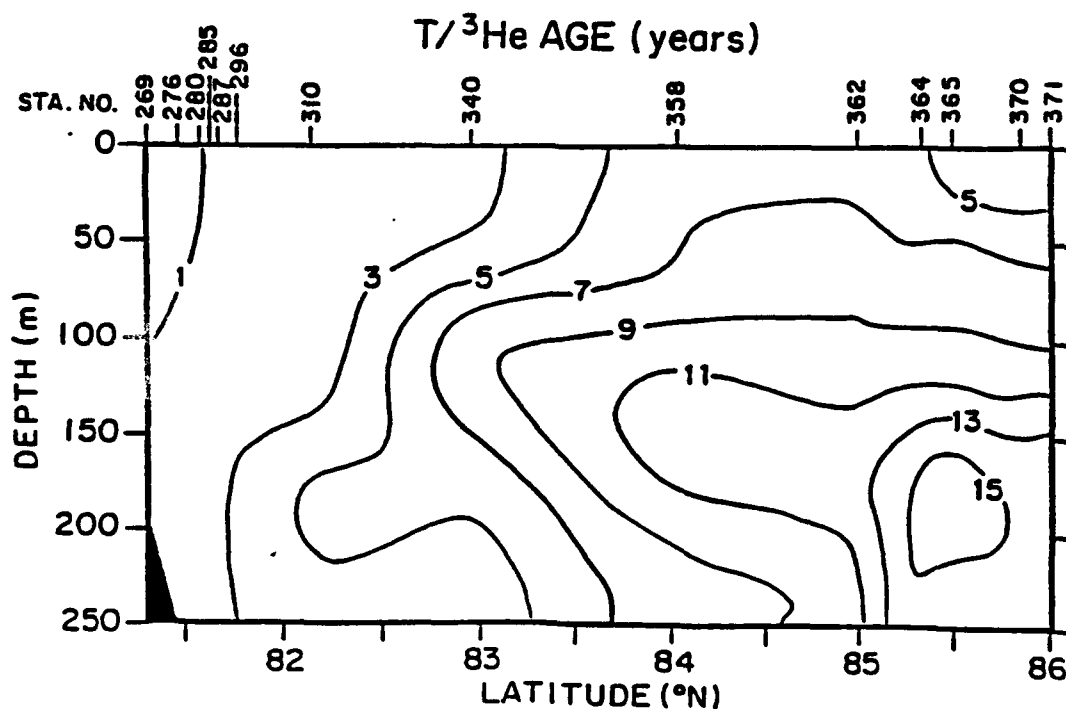


Fig 9: Tritium/<sup>3</sup>He age section of the upper 250 meters for the section extending from the Barents Shelf (sta. 269) to the Gakkel Ridge (Sta. 371; for geographical position of the stations, see Fig. 7). From Schlosser et al., 1993.

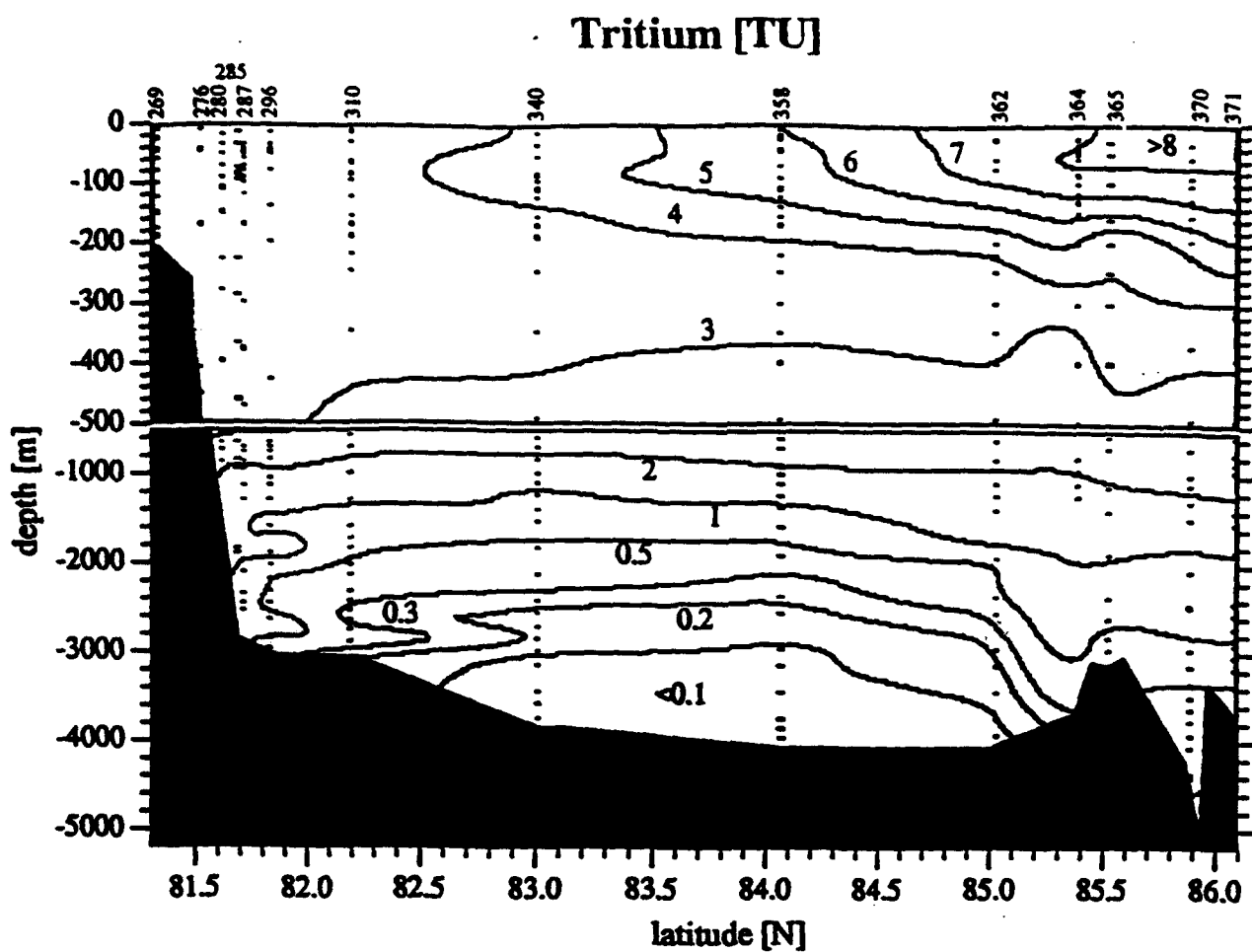


Fig. 10 Tritium section across the Nansen Basin (for geographical positions of the stations, see Fig. 7; Boenisch and Schlosser, unpublished data).



# Mobility of Radioisotopes in Marine Surface Sediments

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## ABSTRACT

*Transport of a radioisotope in a sediment-water system can be retarded by sorption of the isotope to solids, which is controlled by the affinity of the radioisotope for the sediment particles. In order to study trace metal and radionuclide mobility on the sea floor, the following measurements were carried out: (1) effective diffusion rates in sediments in the laboratory and on the sea floor, (2) laboratory distribution ratio ( $K_d$ ) values, which measure the affinity of an element for the solid phase, and (3) field  $K_d$  values. Effective diffusion rates and  $K_d$  values for the radioisotopes  $^{134}\text{Cs}$ ,  $^{125}\text{Sb}$ ,  $^{203}\text{Hg}$ ,  $^{133}\text{Ba}$ ,  $^7\text{Be}$ ,  $^{65}\text{Zn}$ ,  $^{54}\text{Mn}$ ,  $^{60}\text{Co}$ ,  $^{59}\text{Fe}$ , and  $^{113}\text{Sn}$  were measured for six marine sediments: carbonate ooze, siliceous clay, red clay, metalliferous sediment, hemipelagic sediment, and terrigenous clay. The terrigenous clays analyzed are similar in composition and texture to sediments from sites of present and potential radioactive waste contamination in the arctic.*

*Based on the effective diffusion coefficients ( $D_s$ ) determined in the laboratory, the isotope mobility fell into five groups. Diffusive transport rates fall into similar mobility groups for all sediment types tested both in the laboratory and on the sea floor. Interaction of the radionuclide with solids was the primary mechanism for reducing its mobility in the sediment-water system. As it is difficult to measure *in situ* diffusion rates directly, this suggests that  $K_d$  values based on measured sediment properties can be used to help determine *in situ* diffusion rates for radioactive waste near the sediment-water interface for other locations, such as the Arctic. The relative contributions of diffusion, bioturbation, irrigation, and sediment transport to radionuclide transport can then be assessed and used to predict the interactions and fate of the radioactive waste in the sedimentary environment.*

## INTRODUCTION

Once a radioactive substance is released by a waste source, its fate and the scope of its impact are dependent on the interactions that occur with the environment. Contaminants partition between dissolved and particulate phases according to their inherent chemical properties and are dispersed or concentrated by natural processes such as water circulation, dilution, evaporation, sediment transport, resuspension, and biological or inorganic fixation processes. For many nuclides, the sediments can effectively isolate the waste from the biological cycles, thereby reducing adverse impacts to the ecosystem or to people. An understanding of the interaction of radionuclides with the sediments is necessary to determine the rates and mechanisms by which wastes move in the sedimentary environment. Reasonable predictions of the fate of wastes can then be made, and appropriate remediation undertaken. The radioisotope mobility studies that are presented here were initially undertaken as part of the Manganese Nodule Program (MANOP) to study the transfer of metals and other trace elements across the sediment-water interface and in the surficial sediments of the Pacific Ocean. The results of these studies show that the type and degree of interaction that artificial radionuclides have with marine sediments follows patterns that are often predictable and are also applicable to other locations. These patterns provide insights for assessing the propensity of radioactive waste to move in an environment, and its coastal waters, where weapons production and reactor wastes have been dumped or may be released by future accidents. The Arctic, which is the focus of this conference, is one such environment where questions have been raised about potential environmental hazard from dumped waste in the Kara sea, sunken nuclear submarines, and radioactive releases through the watersheds.

Diffusion in the sediment system is one mode of dispersion for a chemical or radioisotope.



Diffusive transport of a radioisotope in a sediment-water system can be retarded by sorption of the isotope to solids. The degree of sorption is controlled by the affinity of the radioisotope for the sediment particles; it is very dependent on the mechanism of sorption or reaction and on the composition of the sediment. The distribution ratio,  $K_d$ , is often used to quantify the partitioning between solid and solution and to predict the retardation of diffusive transport due to sorptive processes. It is defined as the concentration of an entity (radioisotope in this case) in the solid phase relative to that in the aqueous phase. We measured  $K_d$  values for a suite of radionuclides and sediments in laboratory batch experiments to compare measured diffusive transport rates with those predicted from measured  $K_d$  values. A final test of the predictive value of the laboratory measurements was made by comparison with in situ  $K_d$  values and relative transport rates that were determined on the sea floor.

## METHODS

In order to study radionuclide and trace metal mobility on the sea floor, the following measurements were carried out (details presented in Buchholtz ten Brink, 1987):

- (1) effective diffusion rates in sediments in the laboratory (Figure 1a-c)
- (2) laboratory  $K_d$  values (Figure 1d)
- (3) molecular diffusion rates in seawater in the laboratory
- (4) field  $K_d$  values (Figure 1e)
- (5) radioisotope transport rates in sediments on the sea floor (Figure 1e)

Measured diffusive transport rates were then compared with those predicted from measured  $K_d$  values.

Six marine sediments representing a broad compositional range (Table 1) and a suite of 14 radioisotopes (Table 2) that encompassed a variety of chemical properties were used in the experiments (Table 3). Sediments and in situ experiments were from the MANOP Lander program, however, the behavior observed for the radioisotopes can be extrapolated to sediments from other sites of potential or real contamination by radioisotopes or metals. The bulk composition of the sediments was determined for representative samples used in the various experiments and properties that influence the solid-solution behavior, such as the surface area of the sediment particles and their cation exchange capacity, were also determined. The terrigenous clays analyzed are similar in composition and texture to sediments from sites of present and potential radioactive waste contamination in the arctic. The isotopes studied were gamma-emitters and the concentration of all isotopes was determined simultaneously for each sample by detection on a Li-drifted germanium gamma detector. Separation of solids from the aqueous phase, which is necessary to obtain  $K_d$  values, was done by either filtration or centrifugation.  $K_d$  was then calculated by dividing the concentration of radioisotope in the solid phase by the radioisotope concentration in the aqueous phase. The equation used to calculate the effective diffusion rate for nuclides in the sediment from values of parameters measured in the laboratory is:

$$D_s = D_{sw} / \{ (T^2) (1 + (K_d \cdot \rho_s \cdot (1 - \phi) / \phi)) \} \quad (1)$$

when bioturbation and sedimentation terms are small and

$D_s$  = effective diffusion coefficient in the sediment-porewater system ( $\text{cm}^2/\text{s}$ )

$D_{sw}$  = molecular diffusion coefficient in seawater ( $\text{cm}^2/\text{s}$ )

$K_d$  = the distribution ratio; quantifies the partitioning between solid and solution due to sorption ( $\text{g/ml}$ ) = (conc. on solid)/(conc. in solution)

$T$  = tortuosity, function of  $\phi$  (unitless)

$\rho_s$  = sediment grain density

$\phi$  = porosity

These parameters are considered to be constant with depth in the sediment for simplicity in using equation (1), however, all of them do actually vary down core to differing degrees. The effect that such variations have on diffusivity will only be discussed briefly here (see Buchholtz ten Brink, 1987).

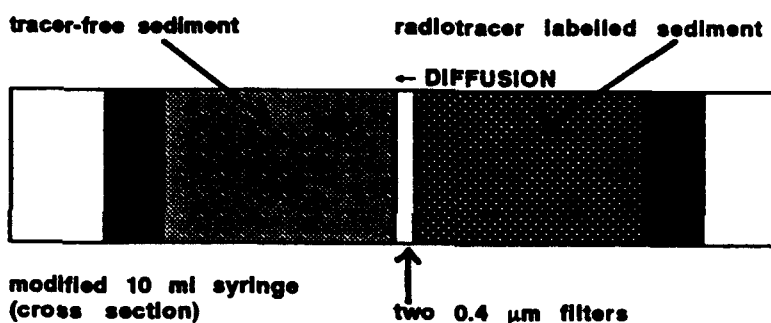
Figure 1. Schematic of experimental procedures used to determine the  $K_d$  value and to measure the diffusive mobility of radioisotopes in marine sediments. Techniques used include (a) sediment-sediment plug technique (Li and Gregory, 1974), (b) sediment-agar plug technique, (c) intact core technique, (d) batch slurry equilibration (Duursma and Bosch, 1970; Li et al., 1984, Nyffeler et al., 1984), and (e) *in situ* equilibration on the sea floor (Santschi et al., 1984). Additional details of the procedures are given in the references cited and in Buchholtz ten Brink, 1987.

## Measurement of $D_s$ and $K_d$ in the laboratory

### a) Homogenized sediment-homogenized sediment

$4^\circ \pm 2^\circ \text{ C}$ ,  $t = 92$  days  
each half gamma counted for radioisotopes  
and  $D_s$  calculated from  $D_s = \pi^2 x^2 / t$

each half also centrifuged, separated, and gamma counted  
to obtain  $K_d$  values



### b) Homogenized sediment-seawater (with agar)

$4^\circ \pm 2^\circ \text{ C}$ ,  $t = 270$  days  
sectioned into 2 mm intervals,  
each interval gamma counted for radioisotopes,  
radiotracer concentration determined as a function of depth

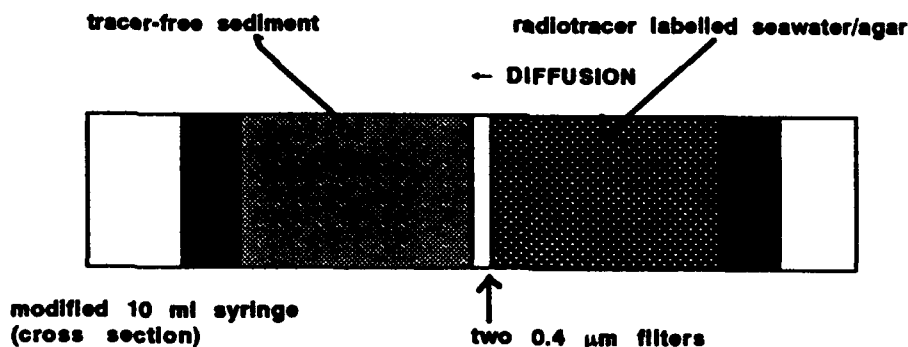


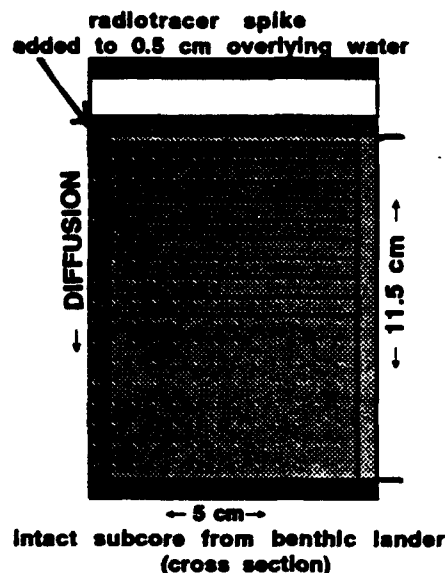
Figure 1 continued.

c) Intact subcores

$4^\circ \pm 2^\circ \text{C}$ ,  $t = 135$  days  
 sliced in 2 mm intervals,  
 each interval trimmed and  
 gamma counted, radiotracer  
 concentration determined as  
 a function of depth

$D_s$  calculated from  $\ln (C_0/C) = z^2/4D_s t$

upper 1.5 cm samples also  
 centrifuged to obtain  $K_d$  values

d) Batch  $K_d$ 

$4^\circ \pm 2^\circ \text{C}$ ,  $t = 0-60$  days  
 particle concentration 0.8 g/l in 1 liter polyethylene bottle  
 sediment slurry added to radioisotope-labelled seawater  
 filtered, sediment and water gamma counted

Calculate  $K_d = (\text{conc. on solid})/(\text{conc. in solution})$

e) Measurement of  $D_s$  and  $K_d$  in situ

$\sim 4^\circ \text{C}$ ,  $t = 1$  to 4 days  
 subcores sliced in 2-3 mm intervals,  
 sediment gamma counted,  
 radiotracer concentration determined as a function of depth

suspended & surface sediment also filtered  
 or centrifuged to obtain  $K_d$  values

*Schematic cross section of Benthic chamber and a subcore removed from it*

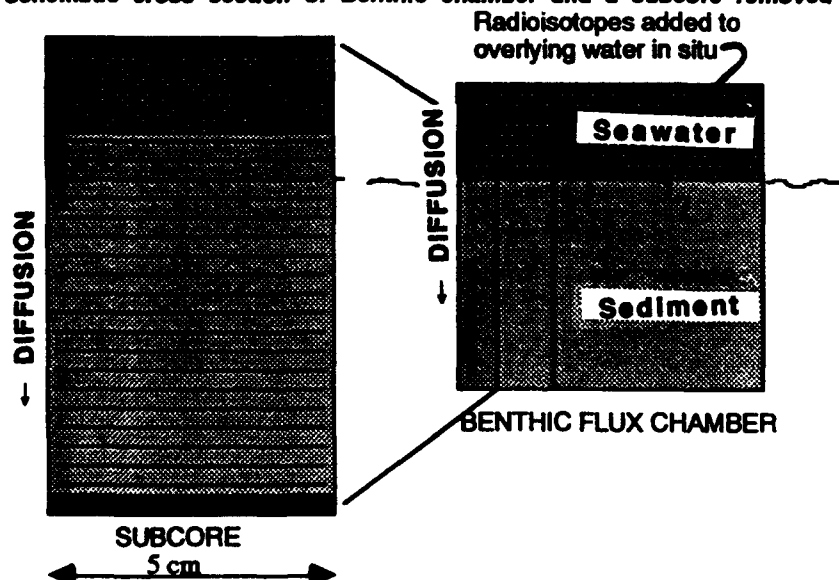


Table 1. Composition of the sediments used for the laboratory radiotracer mobility experiments. Values for Kara Sea sediments\* (Kulikov, 1961) are given for comparison.

### Composition and properties of the sediments

Sediment type	Source	Depth (cm)	Corg (wt%)	CaCO <sub>3</sub> (wt%)	Si (wt%)	Al (wt%)	Fe (wt%)	Mn (wt%)	Surface area (m <sup>2</sup> /g)	Cation exchange capacity (moles/kg)
carbonate ooze	MANOP Site C	0-0.2	0.5	76	6.5	0.51	0.37	0.18	23	0.4
	MANOP Site C	1.8-2.0	0.3	76	6.5	0.57	0.37	0.18	23	0.4
siliceous clay	MANOP Site S	0-0.2	1.3	0.1	27	5.9	4	0.25	31	1
	MANOP Site S	1.8-2.0	0.5	0.1	27	5.9	4	0.25	31	1
terrigenous clay/ continental shelf	San Clemente Basin (Site L)	0-2	2.5	14	20	2.4	6.2	1.43	1.2	0.5
	(Site L)	5.9-6.1		17	20	2.4	6.2	0.21	1.2	0.5
metalliferous	MANOP Site M	0-5	1.3	14	23	6.2	8	2	58	1
hemipelagic	MANOP Site H	0-5	0.7	1.5	24	6.1	5.1	5.4	63	1.4
red clay	MANOP Site R	0-5	0.2	0.6	25	9.1	6.7	0.3	46	0.6
terrigenous mud/clay	Kara Sea*	surface	0.5-1	1			2-5	.02-.6		

The composition of the bulk sediments at MANOP sites was determined by M. Lyle at Oregon State Univ., generally by X-ray fluorescence, and that of the Site L Sediments by D. Hammond (at USC), D. Heggie (at URI), P. Santschi (at LDGO) and U. Nyffeler (at LDGO). Organic carbon values are from Emerson et al. (1985) and C. Reimers (SIO). B.E.T. surface area was determined by U. Nyffeler at U. Bern and the cation exchange capacity at LDGO (by M. Buchholtz) using the tritium method of Yeates and Healy (1976). Sediment particles from the laboratory experiments were further examined with scanning electron microscopy (SEM) to observe individual particle size distribution and composition.

Table 2. Radioisotopes used in the experiments and their properties. All are gamma-emitting nuclides that were added at trace levels to seawater or sediment in the chemical form noted.

ISOTOPE	MAJOR PEAK (KEV)	HALF LIFE	CHEMICAL FORM ADDED <sup>1</sup>	MAJOR SPECIES IN SEAWATER <sup>2</sup>
<sup>22</sup> Na	1274.5	2.602 y	NaCl	Na <sup>+</sup>
<sup>134</sup> Cs	604.7	2.062 y	CsCl	Cs <sup>+</sup>
<sup>133</sup> Ba	356.0	10.66 y	BaCl <sub>2</sub>	Ba <sup>2+</sup>
<sup>125</sup> Sb <sup>3</sup>	428.0	2.71 y	SbCl <sub>3</sub> , SbCl <sub>5</sub>	Sb(OH) <sub>5</sub> , Sb(OH) <sub>6</sub> <sup>-</sup>
<sup>65</sup> Zn	1115.5	244.0 d	ZnCl <sub>2</sub>	Zn <sup>2+</sup> , Zn(OH) <sup>+</sup> , ZnCO <sub>3</sub>
<sup>7</sup> Be	477.6	53.3 d	BeCl <sub>2</sub>	Be(OH) <sub>2</sub>
<sup>203</sup> Hg	279.0	46.8 d	Hg(NO <sub>3</sub> ) <sub>2</sub>	HgCl <sub>2</sub> , HgCl <sub>4</sub> <sup>2-</sup>
<sup>54</sup> Mn	834.8	312.2 d	MnCl <sub>2</sub>	Mn <sup>2+</sup> , MnCl <sup>+</sup>
<sup>60</sup> Co	1173.2	5.3 y	CoCl <sub>2</sub>	Co <sup>2+</sup>
<sup>59</sup> Fe	1099.2	44. d	FeCl <sub>3</sub>	Fe(OH) <sub>3</sub> , Fe(OH) <sub>2</sub> <sup>+</sup>
<sup>113</sup> Sn	391.7	115.1 d	SnCl <sub>4</sub>	Sn(OH) <sub>4</sub> , Sn(OH) <sub>3</sub> <sup>+</sup>
<sup>153</sup> Gd	97.5	241.6 d	GdCl <sub>3</sub>	Gd(OH) <sub>3</sub>
<sup>139</sup> Ce	165	137.2 d	CeCl <sub>3</sub>	Ce(OH) <sub>3</sub>
<sup>141</sup> Ce	145	32.5 d	CeCl <sub>3</sub>	Ce(OH) <sub>3</sub>

<sup>1</sup> According to manufacturers' specifications

<sup>2</sup> Duursma and Eisma (1974), Baes and Mesmer (1976), Stumm and Morgan (1981)

<sup>3</sup> Manufacturer indicated that Sb was a mixture of Sb(III) and Sb(V) in unknown proportions.

Table 3. Matrix indicating which experiments were done at each site and for each sediment type.

Experimental matrix								
	In situ transport ( $\leq 4.5$ d)	In situ $K_d$	Lab $D_{sw}$	Lab $K_d$ batch time series (0-60 d)	Lab $K_d$ natural porosity (92 d)	Lab $D_s$ sed-sed plug (92 d)	Lab $D_s$ agar-sed plug (270 d)	Lab $D_s$ intact subcore (135 d)
<b>SEDIMENT TYPE</b>								
carbonate ooze	✓	✓	✓	✓	✓	✓	✓	
siliceous clay	✓	✓	✓	✓	✓	✓	✓	
red clay			✓	✓	✓	✓	✓	
metalliferous	✓		✓	✓	✓	✓	✓	
hemipelagic	✓		✓	✓	✓	✓	✓	
terrigenous/continental shelf	✓	✓	✓	✓	✓	✓	✓	✓

The diffusivity was measured directly in the laboratory by determining the rate and distance of radioisotope movement from a region of high concentration (radio-labelled sediment, seawater or seawater gel) into sediments that initially had no artificial radioactivity. The concentration of radioisotope in each sediment, gel, or water sample was determined by gamma counting. The effective diffusion coefficient was calculated from solutions to diffusive transport models using the boundary conditions appropriate for each specific experimental setup (Figure 1).  $D_s$  values for in situ conditions were obtained by best fits to a numerical model (Nyffeler et al., 1984, 1986) which allowed downcore variations in some parameters.

## RESULTS

Measured mobility varied greatly between isotopes, but the effective diffusion coefficients ( $D_s$ ) for radioisotopes in the laboratory fell into distinct groups (Table 4) for the Site L continental shelf sediments, which are compositionally and texturally similar to sediments from the Arctic waste sites (Table 1).

Table 4. Effective diffusion rates for radioisotopes measured in clay sediments from the continental shelf (Site L in San Clemente Basin, CA) as determined by the sediment plug technique (Fig. 1a). Values are given in order of decreasing mobility.

ISOTOPES	$D_s$ ( $cm^2/s$ )
(I) $^{22}Na$	$10^{-6}$
(II) $^{134}Cs$ and $^7Be$	$4.9 \times 10^{-8}$
(III) $^{125}Sb$ , $^{133}Ba$ , and $^{203}Hg$	$1.4 \times 10^{-8}$
(IV) $^{65}Zn$ (and $^7Be^*$ )	$10^{-8} - 10^{-9}$
(V) $^{54}Mn$ and $^{60}Co$	$10^{-9}$
(VI) $^{59}Fe$ , $^{113}Sn$ , $^{153}Gd$ , $^{139}Ce$	$< 10^{-10}$

\* $^7Be$  mobility was more similar to that of  $^{65}Zn$  for other sediments tested and its behavior for site L was not consistent.

The effective diffusion rate of the isotopes in the sediments for the radioisotopes was similar in both the laboratory mobility experiments of 53 -135 days and the in situ experiments of 1-4 days. The relative mobility of the isotopes is illustrated by diffusion profiles measured with the intact core technique for sediments from San Clemente Basin (Figure 2).  $^{22}\text{Na}$  diffusion was so much greater than the other isotopes that it was present uniformly throughout the core. Diffusive transport rates fall into similar groups for all sediment types tested, although differences in sediment composition and characteristics (Table 1) produced a range of  $K_d$  values (Table 5) and hence  $D_s$  values for a given isotope (Figure 3).  $D_{sw}$  is ~constant from one site to another while  $K_d$  values for some isotopes vary by up to a thousand fold ( $10^3$ ) between sediment types. The decreases in measured diffusive transport into the sediments are consistent with measured increases in  $K_d$  values. A straight line in the 'concentration (or normalized inventory) vs. depth squared' profiles (Figure 2a and Figure 3) indicates that boundary conditions for equation (1) are met. The curvature below the surface in the intact subcore (Figure 2b) suggests that the  $K_d$  value is decreasing with depth in the sediment. Changes in solids composition down core (Table 1), related to changing redox conditions and carbon utilization, created these variations in  $K_d$  (example in Table 5b) and diffusivity with depth.

$D_s$  values for the six sediments studied could be predicted well (better than two orders of magnitude, depending on the isotope) from equation (1) when the appropriate laboratory  $K_d$  value was used (Buchholtz ten Brink, 1987). This suggests that predictions of radioisotope mobility in other sediments that are made based on either laboratory measurements of diffusivity or measurement of the sediment composition will also be sufficiently accurate for assessments of environmental safety. Table 6 compares  $D_s$  values measured in the laboratory with those predicted from  $K_d$  values measured on the same sediments. Generalizations can be made about the reliability of such predictions by considering the in situ mobility and  $K_d$  values along with the laboratory values for all sediment types (see Buchholtz et al., 1985).

Table 6. Comparison of predicted and measured values of diffusivity for continental shelf sediments from San Clemente Basin, CA. The accuracy of predictions for these isotopes for sediments from other sites is similar to the examples given here.

Continental Shelf Sediment (Site L)						
	Homogenized sediment			Intact core		
	$D_s$ ( $\times 10^{-8} \text{ cm}^2/\text{s}$ ) 92 days, $\sigma = .85$			$D_s$ ( $\times 10^{-8} \text{ cm}^2/\text{s}$ ) 135 days $\sigma = .68-.92$		
	measured	predicted	ratio	measured	predicted	ratio
$^{133}\text{Ba}$	2.06	4.18	0.5	3.13	2.99	1
$^7\text{Be}$	1.34	1.75	0.8	5.18	0.14	37
$^{60}\text{Co}$	0.225	2.5	0.1	0.232	32.5	0
$^{137}\text{Cs}$	1.54	4.32	0.4	4.63	8.1	0.6
$^{203}\text{Hg}$	1.83	2.65	0.7	2.99	2.35	1.3
$^{54}\text{Mn}$	11.7	0.12	98	0.48	-	-
$^{125}\text{Sb}$	4.19	4.04	1	0.99	4.32	0.2
$^{65}\text{Zn}$	0.092	0.21	0.4	0.22	0.14	1.6
$^{113}\text{Sn}^*$	0.00044	-	-	$\leq 0.065$	-	-
$^{59}\text{Fe}^*$	0.00064	-	-	$\leq 0.067$	-	-

*D<sub>s</sub> values are predicted from K<sub>d</sub> values measured at in situ porosities.  
K<sub>d</sub> values were also measured at other porosities and yield similar results.  
\* no K<sub>d</sub> values could be measured at in situ porosities.*

Figure 2. a) Diffusion profiles at the end of the 135 day test for selected radioisotopes in an intact subcore from the continental shelf (Site L). Radioisotopes were added to overlying water and allowed to diffuse into the sediment, then the subcore was sectioned and the isotope concentrations measured in the sediment. Concentrations are normalized (area under normalized inventory curve is in all cases equal to 1) so that tracers can be compared to each other despite differing amounts of radioisotope added. Deeper penetration and lower surface values occur for faster diffusive transport. b)  $D_s$  ( $\text{cm}^2/\text{s}$ ) values are at right; when profiles are non-linear, values are derived from the upper portion.

#### Diffusion in the lab for Continental Shelf Sediments

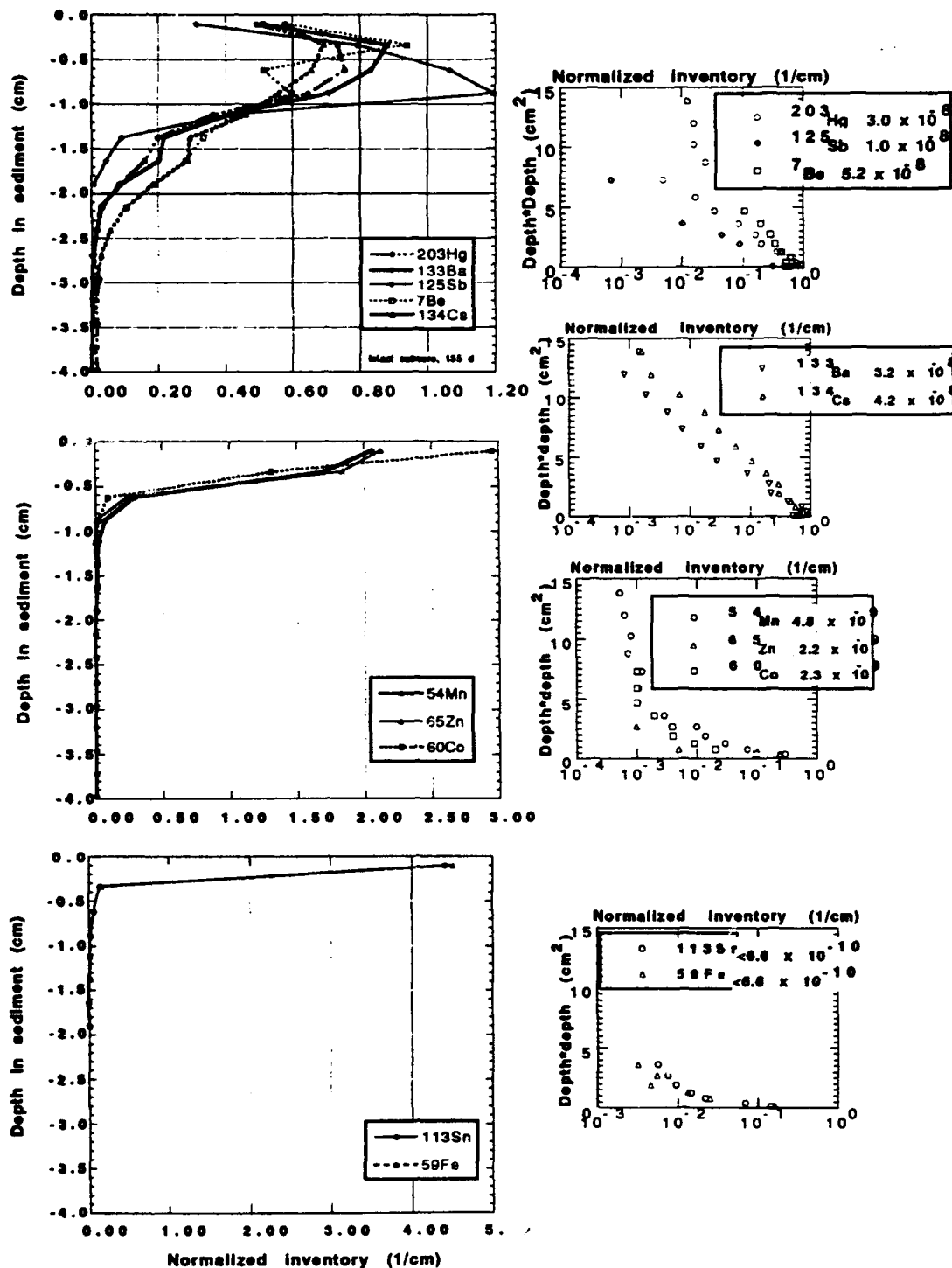


Table 5. The effect that differences in sediment composition have on sorption and hence diffusion can be seen by differences in the  $K_d$  values for sediments with differing compositions (see also Santschi et al., 1990). (a) Equilibrium (30 days)  $K_d$  values measured with the batch equilibration technique (Fig. 1d) for each of the isotopes and each of the sediment types (from Buchholtz et al., 1986) for a sediment concentration of 0.8 mg/l. An additional 30 days equilibration usually produced no significant change. (b)  $K_d$  values measured in the laboratory at in situ porosities (Fig. 1a, 1c). Many isotopes (especially those of Fe, Sn, and Ce; not shown) will have lower  $K_d$  values at higher particle concentrations.

a) Equilibrium distribution ratio ( $K_d$ ) values for Batch method

Sediment 0.8 g/l	$^{134}\text{Cs}$	$^{133}\text{Ba}$	$^{125}\text{Sb}$	$^{65}\text{Zn}$	$^{60}\text{Co}$	$^{54}\text{Mn}$	$^{203}\text{Hg}$	$^{59}\text{Fe}$	$^{139}\text{Ce}$	$^{113}\text{Sn}$
C1	80	515	515	2,070	174,000	17,500	20,900	480,000	486,000	371,000
C2	86	530	490	1,880	64,200	4,980	10,400	667,000	359,000	552,000
S1	711	560	463	832	32,800	1,180	2,960	195,000	251,000	240,000
S2	693	510	507	992	64,600	2,350	2,940	200,000	98,000	200,000
L1	570	99	108	3,210	299,000	251,000	62,000	507,000	330,000	405,000
L2	550	122	143	5,010	448,000	550,000	17,900	610,000	366,000	492,000
M	409	2,540	1,740	8,330	510,000	629,000	3,760	280,000	570,000	360,000
H	390	3,920	8,330	52,300	1,110,000	2,450,000	2,430	760,000	2,000,000	500,000
R	600	500	110	1,010	300,000	20,000	-	-	-	-

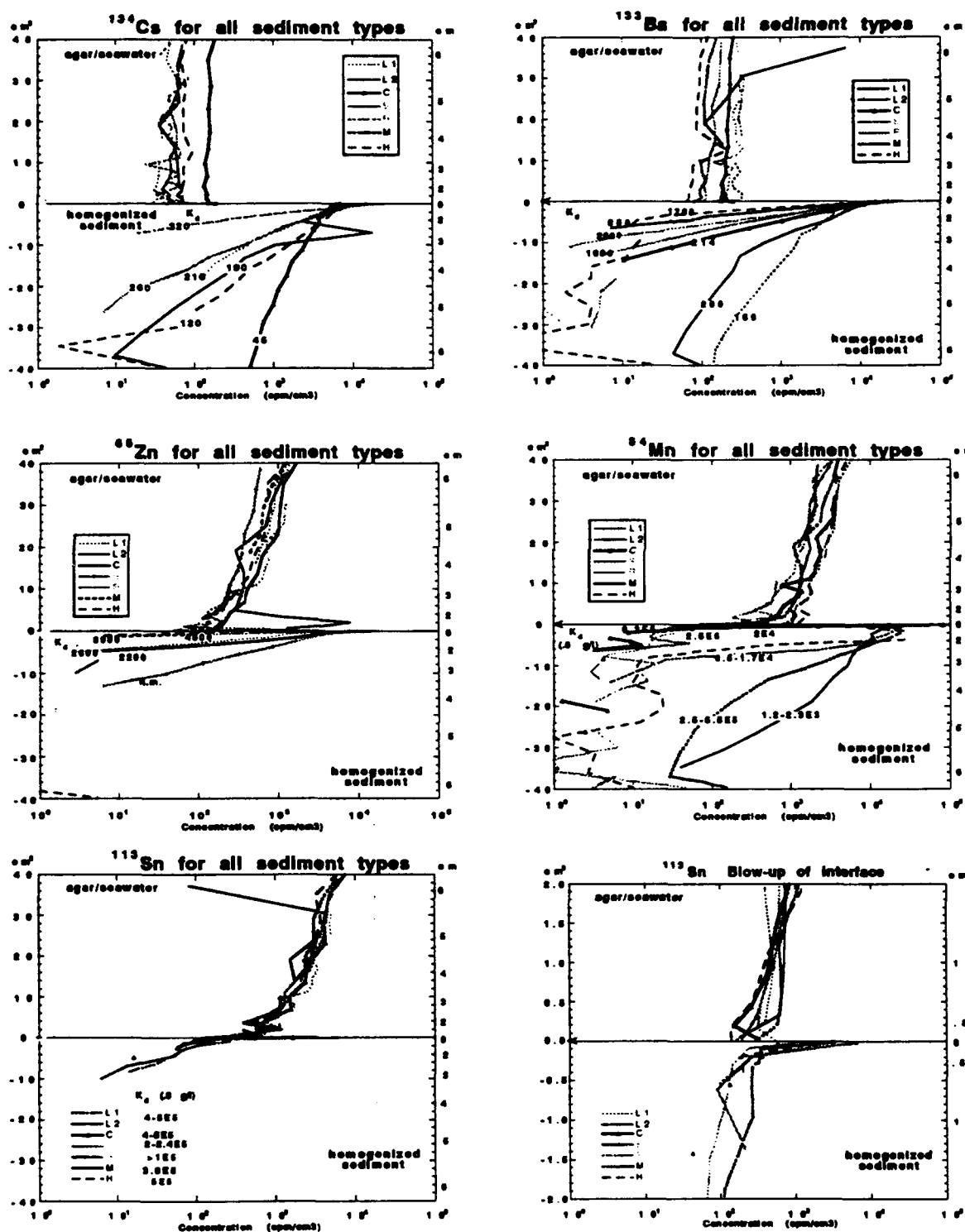
5b) Distribution ratio values ( $K_d$ ) measured in the laboratory at in situ porosities

Sediment	Depth in core (cm)	Porosity	Equilibration time (d)	Distribution ratio $K_d$ (ml/g) i.e. (cpm/g sediment)/(cpm/ml water)							
				$^{134}\text{Cs}$	$^{133}\text{Ba}$	$^{125}\text{Sb}$	$^7\text{Be}$	$^{65}\text{Zn}$	$^{60}\text{Co}$	$^{54}\text{Mn}$	$^{203}\text{Hg}$
Site M	0-5	0.85	49	210	850	600	110	-	1130	-	320
Site H	0-5	0.88	49	120	1700	4200	180	8900	1550	-	500
Site R	0-5	0.70	49	320	200	140	-	930	500	2300	180
Site S	0-2	0.87	49	260	1090	820	-	0	800	6100	490
Site C	0-2	0.76	49	45	214	300	43	4000	580	-	170
Site L a	0-2	0.84	49	190	75	25	-	2200	1230	-	76
Site L b	0-2.5	0.85	49	190	200	430	180	-	1980	-	190
Site L c	0-2	0.85	150	370	160	140	-	2600	220	3900	180
Errors in $K_d$ due to counting uncertainty (%)				2	2	5	15	7-40	2	10	10
Site L d	0.0-0.2	0.92	≤135	345	400	230	3900	7100	1E+05	-	350
	0.2-0.4	0.92		370	700	450	-	-	36000	-	650
	0.4-0.6	0.91		270	550	880	-	-	-	-	180
	0.6-0.8	0.91		400	760	2700	-	-	-	-	10
	0.8-1.2	0.90		410	690	2400	-	-	-	-	65
	1.2-1.4	0.90		390	590	-	-	-	-	-	40
	1.4-1.6	0.89		330	420	-	-	-	-	-	20
	1.6-1.8	0.88		320	260	-	-	-	-	-	15
Errors in $K_d$ due to counting uncertainty (%)				2	2	5	40	7	20	-	10

Homogenized sediment from a MANOP Lander or Soutar Box core was used except for Samples Lc and Ld. Sample Lc was homogenized from a subcore of a Lander boxcore, and Ld is an intact Lander subcore.



Figure 3. Profiles of radioisotope concentration in six different sediments (agar-sediment plug technique) showing the variation in diffusion rates due to compositional changes.  $K_d$  (ml/g) values, which parameterize the sediment-isotope interaction, are given near the corresponding profile. Depth scale is  $\text{cm}^2$  so that linear regions indicate constant  $D_s$  and  $K_d$  in the homogenized sediment. Sample interval of 2 mm is shown with symbols for selected sites.



Diffusion rates for  $^{134}\text{Cs}$ ,  $^{125}\text{Sb}$ ,  $^{203}\text{Hg}$  and  $^{133}\text{Ba}$  could be predicted within a factor of two when  $K_d$  was determined on the same sediment at natural porosity.  $^7\text{Be}$  behavior in the San Clemente Basin shelf sediments was similar to that of  $^{134}\text{Cs}$ ; however, for all the other sediment types tested, its mobility was more similar to that of  $^{65}\text{Zn}$ . For all the sediment types, diffusion rates for  $^{65}\text{Zn}$ ,  $^{54}\text{Mn}$ ,  $^{60}\text{Co}$  could only be predicted to within an order of magnitude when  $K_d$  was determined on the same sediment at natural porosity. The poorer prediction is probably due to a decrease in  $K_d$  within the sample due to local compositional changes, and consequent local decrease in  $D_s$ . In the intact subcore, a decrease in the amount of solid manganese with depth in the core produced a decrease in  $K_d$  with depth. Local variations of redox conditions may have affected the  $K_d$  in other cases also.  $D_s$  for  $^{59}\text{Fe}$  and  $^{113}\text{Sn}$  could not be predicted due to uncertainty in  $K_d$  and  $D_{sw}$ . The  $K_d$  values measured in the laboratory were most strongly a function of the particle concentration and, because of the high affinity of these isotopes for solids (Table 4), the sampling resolution was not always adequate to determine accurate values at in situ porosities. In addition, the low values measured for diffusion of Fe and Sn in the unfiltered seawater suggested that these nuclides were present in solution in a colloidal or macromolecular state (Buchholtz ten Brink, 1987).

Sorptive behavior for nuclides on natural sediment, expressed by the  $K_d$  value for a particular sediment, can be correlated to specific sediment and chemical properties (Buchholtz, et al., 1985; Balistrieri and Murray, 1986). Studies of the partitioning between sediment and seawater showed that:

- 1) equilibrium  $K_d$  for most isotopes studied is controlled primarily by sediment properties:

<i>Isotope</i>	<i>Controlling property</i>
$^{54}\text{Mn}$ , $^{60}\text{Co}$ , $^{65}\text{Zn}$ , $^{133}\text{Ba}$ , $^{125}\text{Sb}$ , $^{141}\text{Ce}$	$K_d \propto \% \text{Mn}$
$^{133}\text{Ba}$ , $^{125}\text{Sb}$	$K_d \propto \% \text{ surface area, ion exchange capacity}$
$^{134}\text{Cs}$	$K_d \propto \% \text{ Si, clay content, } 1/\text{CaCO}_3$

- 2) kinetics of equilibrium are primarily determined by uptake mechanism

- 3) particle concentration effects are important for Fe and Sn.

Consequently, as reliable values for sediment characteristics near Arctic waste sites become available, better estimates can be made of the potential for diffusion in the sediments.

Diffusivity in the sediments is only one component of mobility in the environment. The partition coefficient provides a measure of how much of any isotope or material put into the oceans will remain in solution and how much will become associated with the solid phase when contact with the sediments occurs. This contact may be as suspended sediments scavenging radioisotopes from solution in the water column and eventually sinking to the sea floor, it may be direct removal from solution to bottom sediments, or it may be equilibration between radioactive waste solids and bottom sediments. Although kinetic factors and temporal or spatial changes in sedimentary conditions modify any average behavior, the diffusion coefficient in the sediments, whether measured directly or derived, can predict the rate of dispersal of a contaminant in immobile sediments. Most surficial sediments on the seafloor, however, undergo physical advection by bottom currents, biological organisms, and sediment deposition. In the shallow arctic, gouging and resuspension of sediments by ice can also occur. All of the radioisotopes studied have distribution values ( $K_d$ )  $>100 \text{ ml/g}$  (except  $^{134}\text{Cs}$  on carbonate sediments) and prefer a chemically bound or sorbed state over dissolution in the more mobile aqueous phase. Consequently, interaction of the radionuclide with solids is the primary mechanism for reducing isotope mobility in the sediment-water system. The movement of the solid sediments, with their associated radionuclides, can then become a primary mode of transport for radionuclides within both the sediment and the overlying waters.

Comparison of the mobility due to diffusion, bioturbation, sedimentation, and other factors in different environments (Table 7) provides a sensitivity study that can help focus both research and environmental policy. In the deep sea, such as the central Pacific sites of this study, physical mixing processes and sedimentation are usually small compared to the diffusive mobility of isotopes in groups I-III (Table 4) but may be equivalent to or greater than the diffusive mobility of el-

ements with properties similar to those of  $^{59}\text{Fe}$  and  $^{113}\text{Sn}$ . This means that movement of waste-generated nuclides of Cs, Sr, and similar elements in quiescent deep sea conditions can be well approximated by diffusive transport alone. Movement of nuclides of Fe, Sn, Ce, Pu and other very particle-reactive elements will occur primarily by processes that move the sediment itself. In coastal or shelf environments, however, the physical sediment transport processes are likely to dominate mobility for all isotopes that are associated with the sediments. Once a radionuclide becomes associated with the sediment, the rate and scale of resuspension and the strength and duration of bottom currents to transport suspended material from the site will be the predominant factors in radioisotope mobility. Consequently, a realistic assessment of the potential for mobility of radioactive (or other) waste in relatively shallow environments such as the Arctic's Kara sea or Ob estuary requires that rates and processes of sedimentation, resuspension, sediment transport, and biological mixing be well known. Currently, there is great uncertainty in our estimates of sediment resuspension in the Arctic seas; there is also little information available about the sedimentary environment, sediment composition, and biological activity in the local vicinity of existing or potential waste sites. The patterns of radionuclide mobility and other processes in similar environments, however, provide valuable analogues which bound the behavior that is likely to occur in the Arctic.

Table 7. Typical parameters for a suite of isotopes and typical sedimentary environments in the ocean. Mobility of the isotopes in the sediments can be predicted (eqn. 1) and the sensitivity to variations in different terms ascertained.

ISOTOPE	$D_{sw}$ ( $\text{cm}^2/\text{s}$ )	$K_d$ ( $\text{ml/g}$ )	$D_s$ ( $\text{cm}^2/\text{s}$ )	$D_B^*$ ( $\text{cm}^2/\text{s}$ )	$S^{**}$ ( $\text{cm/s}$ )
$^{22}\text{Na}$	$10^5$	1			
$^{134}\text{Cs}$ , $^{125}\text{Sb}$ , $^{203}\text{Hg}$ , $^{133}\text{Ba}$ , $^7\text{Be}$	$3 \cdot 10 \times 10^{-6}$	$10^2$	$10^{-8}$		
$^{65}\text{Zn}$ , $^{54}\text{Mn}$ , $^{60}\text{Co}$	$3 \times 10^{-6}$	$10^3$	$10^{-9}$		
$^{59}\text{Fe}$ , and $^{113}\text{Sn}$	$<3 \times 10^{-6}$ ; $<10^{-9}$	$10^5$	$<10^{-11}$		
<b>Sedimentary Environment</b>					
Deep sea				$10^{-8} - 10^{-11}$	$10^{-11} - 10^{-12}$
Nearshore				$10^{-6} - 10^{-7}$	$10^{-8}$
Kara Sea <sup>#</sup>					$2 \times 10^{-10}$
<b>Physical properties</b>					
$\phi = 0.6-1$ (unitless)					
$T^2 = \phi \cdot (1 \text{ to } 2)$ (unitless)					
$\rho_s = 2.6$ ( $\text{g/cm}^3$ )					
* $D_B$ is the bioturbation coefficient and ** $S$ is the sedimentation rate, typical values are taken from Lerman (1979), Berner (1980), Aller (1982) and Aller and DeMaster (1984)					
<sup>#</sup> from Kulikov (1961)					

## CONCLUSIONS AND IMPLICATIONS

- Equilibrium  $K_d$  values for nuclides of Cs, Ba, Zn, and Hg are most sensitive to sediment composition. Those of Mn, Co, and Sb are strongly affected by redox conditions while Fe, Sn, Ce and Gd are more sensitive to porosity, i.e., particle concentration.
- In situ tests revealed that  $K_d$  values and  $D_s$  values can usually be estimated from laboratory measurements to better than an order of magnitude for a given sediment composition and nuclide.
- As it is difficult to directly measure in situ diffusion rates, this work suggests that for Arctic sediments,  $K_d$  values based on measured sediment properties can be used to help determine in situ diffusion rates for radioactive waste near the sediment-water interface.
- As research continues and the magnitude and variance for each transport mode is better known, the relative contributions of diffusion, bioturbation, irrigation, sedimentation and sediment transport to radionuclide transport can be determined and used to predict the interactions and fate of the radioactive waste in the Arctic sedimentary environment.

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**DESTRUCTION OF THE ECOLOGICAL FOOD-CHAIN IN THE BARENTS SEA  
AS A RESULT OF A COMBINATION OF OVERFISHING AND  
ATMOSPHERIC NUCLEAR TESTING AT NOVAYA ZEMLYA**

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Russian Parliament**

The Barents Sea used to be up to the 1920-1940s one of the most bioproductive basins of the Arctic Ocean.

A combination of warm Gulf stream, influences with a large volume of nitrogen, phosphates and organic substances—from large rivers, as the Pechora for instance—as well as the transport of fertilizers originated from many colonies of fish-eating birds used to be a precious source for the development of phytoplankton, zooplankton and benthos.

The destruction of the ecosystems of the Barents sea is the result of two main processes:

**1. OVERFISHING:**

The USSR and Norway changed the size of their fishing nets and therefore, after collecting adult cods and herrings, they started collecting sub-adults and juvenile specimen also.

## 2. ATMOSPHERIC (AND PROBABLY ALSO UNDERWATER) NUCLEAR TESTING

During the period 1957/1962 more than 80 atmospheric nuclear tests were carried out at the Novaya Zemlya test-site. Most of the testing (1957/61) was carried out in October, but in 1961/1962 the months of August and September were included in the testing period. August and September are exactly the months when fish-eating birds do not migrate with sub-adults from Novaya Zemlya.

Considering that the food-consumption of such colony species as *Uria* for instance is one kg of fish per day, for every specimen, each specimen then produces 28-30 grams of dried guano per day. For 90 days, during the nesting time, this amount equals 2,52/2,70 k kg. Before the start of the nuclear testing programme on Novaya Zemlya, the dimension of the fish-eating birds colony used to be between 4 and 8 million specimen (mostly *Uria*).

Consequently the amount of dried guano produced during a three months nesting period used to be between 10,000.00 tons and 21,600.00 tons. This considerable volume of guano concentrated on the rocks' surface and was distributed to small bays and fjords by the effect of rain, wind and snow. This kind of fertilization was a very important source of phytoplankton production. Phytoplankton is a food source for zooplankton (mostly crustacean *Calanus Finmarchicus*). Zooplankton was also a source of food for juvenile cods (*Gadus*), herrings (*Clupea*) and other fish species.

It has to be considered that no ornithologists visited Novaya Zemlya during the last 30 years. According to recent data (Uspensky and , 1992) at the time being, 30 years after the end of nuclear testing, the density of fish-eating birds is still not restored.

The catastrophic decrease concerning the density of cod population started at the beginning of the 1970s and seems to be a result of overfishing only. But since cods have a life-cycle of 7-10-years (adult specimen can reach the age of 17-19 years), we can suppose that the drastic decrease in the population's density is directly related to the killing of millions of fish-eating birds at the time of atmospheric nuclear

testing—1957–1962. Also, the distribution of guano in the sea water did not take place for at least 2–3 years after the destruction of the bird colonies.

It is indeed difficult to calculate the comparative damage that atmospheric nuclear testing and overfishing had on the Barents Sea ecosystems.

During the last 10–15 years, an additional negative influence can also be found in the effect of oil pollution from the Petchora Sea (causing the destruction of plankton, including planktonic larvae of the fish), as well as in the sonar testing of the ice-surface (???). the combination of such processes contributed to the destruction of the ecosystems of what was once one of the most productive Arctic seas.





**Session 3: Assessment and Remediation**

**Texts Accompanying Presentations**

NON-LOCAL RADIOLOGICAL CONSEQUENCES OF NUCLEAR WASTE  
DUMPING IN THE ARCTIC SEAS: A PRELIMINARY ASSESSMENT

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Abstract

An attempt has been made to predict the long-term radiological impact of radioactive waste disposals in the Arctic Seas on regional and global scales. Based on available information on nuclear waste, including reactors, dumped in the Barents and Kara Seas, a range of source-term scenarios has been studied. A compartmental model has been developed and employed to describe the dispersal of possible radioactive contaminants. This oceanographically sensible conceptual model features a finer spatial resolution in the areas containing sources. Global collective dose commitments have been calculated based on the marine food ingestion pathway. Non-local maximum individual doses associated with the fish ingestion pathway were also evaluated. Dose calculations were performed for unit releases of a range of nuclides into the Barents and Kara Seas and for several complex source-terms, using a range of assumptions leading to conservative predictions. Sensitivity studies are underway.

Provisional modelling results suggest that the global radiological impact of the disposals in the Arctic Seas will be comparable to or less than those resulting from other anthropogenic and natural sources of radioactivity.

IAEA-MEL is also participating in international assessments by providing assistance with direct radioanalytical measurements, intercalibration exercises and collation of a marine radioactivity database.

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## Introduction

This report summarises the recent activities of the IAEA Marine Environment Laboratory in Monaco (IAEA-MEL) in support of the work of the Joint Norwegian-Russian Expert Group and of the Agency's International Arctic Seas Assessment Project (IASAP). The laboratory's assistance covers several aspects, both experimental and theoretical, the primary objective being to provide rapid input to the international collaborations on the practical assessment of the consequences of nuclear waste dumping in the Arctic Seas. As new scientific information becomes available and as opportunities arise to visit and monitor the disposal region, IAEA-MEL is mandated to respond in real time in order to improve the quality and scope of the international assessment. At the present time, because of limited opportunities and information, the IAEA-MEL contribution described hereafter is preliminary in nature and is restricted to considerations beyond the immediate region of waste dumping.

## Inventories and Disposals

In a 1988 IAEA-MEL review of inventories of radionuclides in the oceans, Whitehead noted a 'severe inventory imbalance' for  $^{137}\text{Cs}$  in the Arctic Ocean. A balance between widely based international observations of radionuclide inventory and theoretical predictions of inputs and outputs could be obtained only by postulating additional local source-terms, e.g. close-in fallout from Novaya Zemlya or run-off of contamination from the Urals. The observed mean inventory of  $^{137}\text{Cs}$  for the Arctic, for a ca 1980 reference date, was  $5.5 \text{ GBq km}^{-2}$  with a 10:90% distribution between sediment and water. This inventory would be expected to have decreased subsequently as a result of decreasing inputs, water residence and radioactive decay. A corresponding  $^{137}\text{Cs}$  inventory in the  $2\text{-}3 \text{ GBq km}^{-2}$  region would thus be expected today.

In addition to the radioactivity present in waters and sediments of the Arctic Seas, the disposal of liquid and solid radioactive wastes has increased the total, though not necessarily the available, inventory of radionuclides in these seas. The recent report, White Book 3 (1993) by the Yablokov Committee, provides the first overview of the magnitude of past disposals. 880, 590 and 90,700 TBq of respectively liquid, packaged low/intermediate and solid high-level wastes were disposed of into the marginal Arctic Seas. A subsequent multinuclide assessment by Mount and Schwertz (1993) suggests that  $\sim 10 \text{ PBq } ^{137}\text{Cs}$  were dumped, of which  $\sim 7 \text{ PBq}$  remain. Ratios of actinides: fission products: activation products were approximately 1:50:10. For the subsequent assessment here, the White Book 3 and Mount and Schwertz source-term data have been used. Returning briefly to inventories, however, the White Book 3 implies mean inputs of  $6.6 \text{ GBq km}^{-2}$  of 'available' anthropogenic radioactivity to the waters and sediments of the Barents and Kara Seas (40% global fallout, 9% rivers, 46% Gulf Stream (Sellafield) and 5% liquid waste), with more than 5 times this activity present in packaged/solid waste. Assuming then that  $^{137}\text{Cs}$  contributes a significant fraction to the 'available' total radioactivity, there is good order of magnitude agreement between the White Book model, the Whitehead budget and direct observations, i.e.  $^{137}\text{Cs}$  inventories are in the  $2\text{-}3 \text{ GBq km}^{-2}$  range.

## Radiometric Programme

To assist in the direct measurement of radioactivity in the Kara and Barents Seas, IAEA-MEL participated in the 1992 Joint Norwegian-Russian Expedition. A suite of 7 finely sectioned sediment cores was returned to IAEA-MEL for detailed radiochemical analyses in order to help characterise downcore radioactivity distributions, nuclide activity ratios, inventories, source-terms and sediment chronologies. In addition, a bulk sample of Kara Sea

sediment was returned to IAEA-MEL, freeze-dried, homogenised and distributed to all participating analytical laboratories as an intercalibration exercise associated with IAEA-MEL's ongoing Analytical Quality Control Service. The results of the radiometric programme will, as planned, be presented along with those of the rest of the Norwegian-Russian team in Kirkenes, Norway in August. Preliminary data are not inconsistent with those of the cruise report, showing that radioactive contamination in the Kara Sea is generally very low.

It is anticipated that, in 1993, IAEA-MEL will again contribute to radiometric and intercalibration aspects of the investigation and in addition will deploy a new underwater/seabed gamma-spectrometry system, featuring both NaI and Ge detectors, in order to characterise and map natural and anthropogenic radioactivity distributions in and around the dumpsites.

### Radioactivity Database

The IAEA-MEL Global Marine Radioactivity Database is being developed for all data on marine radioactivity, i.e. sea water, sediments and biota. Information is stored in such a way as to facilitate data interrogation and analysis. It has been implemented at IAEA-MEL to provide a scientific resource designed to serve the following important functions:

1. To provide immediate and up-to-date information on radioactivity levels in the seas and oceans.
2. To provide a snap-shot of activities at any time in any location.
3. To investigate changes with time in radioactivity levels.
4. To identify gaps in available information.

To meet these objectives and to ensure maximum utility of the information contained in the database, the data format has been rigorously prescribed. The degree of detail is extensive (general sample information including type, method of collection and location as well as physical and chemical treatment) to allow the data to be validated and its quality assured. In addition, the database has links to IAEA-MEL's in-house analytical quality control database allowing immediate checks on laboratory practice.

In the specific context of the Arctic Seas, IAEA-MEL has been requested to develop and maintain the database associated with the international assessment (e.g. IASAP). This database will provide crucial evidence in the evaluation of the environmental radioactivity levels of the region and in assessment of the radiation doses to marine biota, local, regional and global human populations. Some of the uses of the database within the Arctic assessment programme are immediate:

1. **evaluation of nuclide ratios** - the identification of the different contributions to radioactivity in the region is critical, given the multiple nature of source-terms, e.g. Sellafield, liquid disposals, leakages from dumped reactors and solid wastes, discharges from the Ob and Yenisey rivers, close-in fallout from testing at Novaya Zemlya, as well as Chernobyl fallout.

2. **Investigation of time trends** - given the temporally varying nature of the known sources to the region, we may be able to estimate their contribution to the environmental concentrations and thus increase the sensitivity with which any small residual change or trend may be detected.
3. **Inventory calculations** - the ability to carry out budget calculations may again permit detection of any imbalances.
4. **dose estimation** - combining the environmental levels with a pathway model will allow estimation (to first order) of the dose to marine biota and local and regional populations.
5. **model validation** - to provide reliable predictions of the impact of real or theoretical discharges, it is necessary to use well validated models and this requires access to the existing appropriate experimental data (either in the form of time series of observations or of a snapshot of activities).

Within the database, we have defined key areas which are actively being researched. Within the Arctic Ocean itself, there are 16 sub-areas (defined by the conventions of the International Hydrographic Organisation). Beyond the confines of the Arctic Ocean, we must also consider those regions of the world's oceans which have significant flows to and from the Arctic (including deep water transport). We are actively seeking data contributions from labs which have been or still are active in these areas. Although the database currently contains around 25,000 data, there are at present only very limited data available for the Arctic Ocean, especially for the Kara and East Arctic Seas. We are therefore actively searching for data and we request contributions from participants in this meeting. The database will be available even at the preliminary stage to all participating institutions and finally to all Member States.

#### The Assessment Model

A wide variety of marine dispersion models exists with differing scales of resolution, parameterisation of physical processes and data requirements. They also differ in their suitability for different functions. Within the Arctic assessment project, we are concerned with local or near and extreme near-field (Kara Sea), regional (Arctic Ocean and associated marginal seas) and global models. We deal here only with the non-local or specifically global radiological consequences of waste dumping, to which end we have chosen to design and implement a number of compartmental models. These models make up one class of marine dispersion models and are particularly suited to long range and long timescale (greater than 100 year) assessments. The accuracy of results from such models is also well suited to the detail required for radiological assessments (NEA, 1985, 1989; CEC 1990). Further, they lend themselves to extensive sensitivity and uncertainty analyses, key components in the evaluation of the reliability of the model predictions.

In compartmental models, the movement of water between various regions is modelled assuming instantaneous and homogeneous mixing within the compartment, flows between adjacent compartments being parameterised by means of rates of transfer. In this way, the processes of advection and diffusion are incorporated. Further significant physical processes can also be included, particularly the scavenging of radionuclides by sediments. This is usually parameterised by means of the distribution coefficient ( $k_d$ ), and such submodels may be incorporated in the near-source regions with the inclusion of further physical

parameters such as sediment load and suspended matter concentration. The effect of ice formation will also be considered in the final model.

The initial assessment work reported here has been carried out on the basis of a preliminary 16 box model, with enhanced structure in the Arctic region (specifically the Kara Sea). A schematic of the model is shown in Figure 1. The basis of this and other models in the series is of course knowledge of the oceanographic and hydrographic structure. In summary, the final model should reflect the relevant oceanography. The Arctic Ocean is almost landlocked and is divided by three submarine ridges into distinct basins. The major inflowing current is the N. Atlantic current which has two arms, the eastward-flowing Norwegian coastal current and the other flowing northwards to the west of Spitsbergen. The Arctic Ocean can then be simplified into three primary components, namely Arctic surface, intermediate and bottom waters. The model used here, MEL ARCTIC2, already has increased detail in the Barents and Kara Sea source-term regions. The Barents Sea, being intermediate between the Atlantic and Arctic Oceans, plays an important role in balancing water transport. Since it has rich fisheries, it also is significant in transporting any leaked radionuclides and their associated dose. The Barents Sea receives Atlantic waters from the west flowing along two branches, the coastal current along the east and a northern flow, while Arctic waters enter between Spitzbergen and Frantz Josef Land and also between Novaya Zemlya and Frantz Josef Land.

The Kara Sea lies entirely on the continental shelf. It is shallow (ave. depth ~120m) with two deep canyons, the Svyataya Anna and Novaya Zemlya (>400m) troughs. The Sea receives a large fresh water input (~1500 km<sup>3</sup>) which forms a seasonal surface layer, Atlantic waters entering from the north, primarily through the Svyataya Anna trough, but also from the west via the Barents Sea. The shallow Laptev Sea is also included in the model. Flows between the model compartments have been taken from the literature and through collaboration with others, notably thus far with I. Harms (Hamburg University) who has specifically developed a 3-D hydrodynamic model for the Barents and Kara Seas, validated from temperature, salinity and Levitus data. The remainder of the model describes the rest of the world's seas, the N. Greenland and Norwegian Seas being explicitly included because of their importance to Arctic flow. The UK coastal system is included because of the potential significance of the Sellafield signal both to Arctic inventories of radionuclides and to testing of the model. The CEC (1990) Marina model was also useful in deriving the south-west section of the MEL ARCTIC2 model.

The model has been carefully balanced for water flow and provides a satisfactorily accurate prediction of Sellafield <sup>137</sup>Cs dispersion through the northern seas. The model also incorporates a radionuclide scavenging removal term for each compartment, this being nuclide-specific and dependent on particle concentrations and fluxes. In addition, a sensitivity analysis is currently being carried out, using the computer codes PREP and SPOP prepared at JRC Ispra.

The oceanographic part of the model produces radionuclide concentration data as output. There follows a radiological component whereby radionuclide concentrations in water are translated into corresponding concentration data in fish, using IAEA-recommended nuclide-specific concentration factor (CF) data. The FAO (1992) and ICES (1992) fisheries catch data are then applied to the compartments of the model so that radionuclide intake into humans can be quantified, assuming that 0.5 of total fish weight is normally consumed, except in the Arctic Seas where a factor of 0.8 is assumed. The final conversion to dose is achieved using ICRP 60 gut-transfer and dose conversion factors. Collective dose commitments are generally truncated to 1000y or less, depending on objective and nuclide half-life.

The foregoing describes a regional and global assessment. In future, it will be possible to address equally important aspects such as local dispersion, radionuclide transfer and dosimetry. With this in mind, collaborations on numerical modelling with I. Harms (University of Hamburg) and V.K. Pavlov (Arctic and Antarctic Research Institute, St. Petersburg) are underway. In addition, review of the ecology of the region has commenced, with collaboration with I.I. Kryshev of SPA Typhoon, Obninsk. The further assistance, in providing local radionuclide data/samples, of S.M. Vakulovsky and A.I. Nikitin, also of SPA Typhoon, of G. Matishov (Murmansk Marine Biology Institute) and of E. Kontar (P.P. Shirshov Institute of Oceanology, Moscow) will hopefully ensure a good local knowledge for input to these local assessments.

A preliminary review of the biology of the area has been carried out at IAEA-MEL (Miquel, 1993; Fowler et al. 1993). Present knowledge on Arctic marine radioecology is, in fact, very limited compared to that gained from temperate and tropical studies. The general lack of hard data involving the processes of radionuclide bioaccumulation and transfer through the food chain in Arctic waters has hampered the validation of similar models based on data generated in temperate latitudes. Therefore, IAEA-MEL has begun to examine the structure of food chains in the Kara and Barents Seas in order to identify key species that would furnish a pathway of radioactivity leading to man. In general, the Kara Sea is poorer in biomass and species diversity than the adjacent Barents Sea. One reason is the fact that ice covers the Kara Sea for nearly three-quarters of the year and hence primary production is severely limited and the links in the food chain are shortened. Benthic biomass in the central Kara Sea and near the eastern coast of Novaya Zemlya where the wastes were dumped ranges from approximately 3 to 10 g m<sup>-2</sup>, whereas in the shallower waters off the Yamal coast of Russia, biomass increases dramatically to 100-300 g m<sup>-2</sup>. The fish population in the Kara Sea is low relative to that in the Barents Sea and, in the central zone of the Kara Sea, fish are generally small and extremely rare. The MEL ARCTIC2 model in fact assumes a fisheries catch for the Kara Sea of 20 tonnes y<sup>-1</sup>, this being located primarily at the mouths and estuaries of the Ob and Yenisey rivers (G. Matishov, pers. comm.). A variety of molluscs, echinoderms, crustaceans and worms form the typical benthic fauna in the region around the dump sites. Of particular interest is that, in the deeper central portion of the sea, the diversity and biomass of molluscs, crustaceans and polychaete worms decrease markedly, with large echinoderms (viz. seastars and brittle stars) becoming the dominant species. Since these echinoderms have very high concentration factors for plutonium and other nuclides, they should be useful bioindicators of contamination and dispersion.

Insights into the possible biological transfer and transport of radioactivity from any contaminated zones in the Arctic environment can perhaps be gained from examining a case study of an accidental plutonium contamination of marine sediments at Thule, Greenland in 1968. Periodic radioecological surveys in the years following the accident indicated that contaminated organisms could be measured out to a distance of 40 km. For shrimp and echinoderms, radioactivity levels did not decrease with distance from the source as rapidly as was found for sediments and infaunal worms and bivalves; this suggests that the more mobile Arctic epifauna can accumulate and transport radionuclides for considerable distances from the contamination source. Yet even in the relatively productive waters near Thule (benthic biomass: 200-300 g m<sup>-2</sup>) several years after the accident, the benthos contained less than 1% of the plutonium inventory in the sediments. Furthermore, no significant increase in plutonium concentration was found in any of the higher animals such as fish, sea birds and marine mammals.



### The Assessment Results

The primary assessment has been carried out on unit activity release to each of the Barents and Kara Seas. Since  $^{137}\text{Cs}$  delivers most of the global collective dose commitment, it is appropriate to begin with a calculation based on a 1 PBq input of  $^{137}\text{Cs}$  to each sea. The results show that 0.5 and 7 man Sv are delivered from unit inputs to the Kara and Barents Seas respectively. For disposal in the Barents Sea, 55% of the dose commitment is transferred via Barents Sea fisheries. For a Kara Sea disposal, the absence of large-scale fisheries generates a much smaller dose commitment, of which 63% is from these limited fisheries in the Kara and Laptev Seas and 25% via the Barents Sea.

Based on the White Book 3 disposal data, the radiological consequences associated with a range of rather extreme scenarios can now be calculated. The results are summarised in Table 1. The maximum collective dose commitment for instantaneous release of all dumped activity is around 150 man Sv, with considerable reductions as more realistic containment times are invoked. Maximum individual dose rates of  $\sim 60 \mu\text{Sv y}^{-1}$  are predicted for Kara Sea fish eaters, again assuming instantaneous release of all disposed activity. More realistic maximum dose rates of  $\sim 1 \mu\text{Sv y}^{-1}$  magnitude are extremely low relative to the 1-5 mSv  $\text{y}^{-1}$  range of natural exposure rates and dose limits.

The development of these radiological consequences follows the time trends for predicted radiocaesium concentrations in seawater shown in Figure 2. For a delayed release of  $^{137}\text{Cs}$  over 20 years, maximum concentrations of  $\sim 10^2 \text{ Bq m}^{-3}$  persist in Kara Sea bottom waters. These are the highest anthropogenic activity concentrations predicted here, yet they remain at <1% of the natural radioactivity of seawater. There are of course marked dilutions and lag-times involved in the temporal trends of  $^{137}\text{Cs}$  concentration in different compartments, with maxima occurring around 30, 35, 55, 60 and 70 years in, respectively, the Barents, N. Greenland, U.K., Central North and Norwegian Seas.

The dose implications associated with other radionuclides have been assessed using the MEL ARCTIC2 model along with the inventory assignments by Mount and Schwertz (1993). Table 2 shows a summary of the conclusions. The activation products dominate the dosimetry of the long-lived nuclides both at time of dumping and 500 years thereafter (500 years being the White Book 3 estimate for the effectiveness of the furfural containment matrix). Globally, the dose generated by the actinides is extremely low ( $<0.1$  man Sv). The largest doses per unit nuclide release ( $10^0$ - $10^3$  man Sv/PBq) are delivered by  $^{14}\text{C}$ ,  $^{60}\text{Co}$ ,  $^{129}\text{I}$ ,  $^{134}\text{I}$ ,  $^{137}\text{Cs}$ ,  $^{99}\text{Tc}$  and  $^{55}\text{Fe}$ . The effect of nuclide scavenging to sediments on resulting dosimetry is particularly noticeable for the activation products and actinides, with typical reductions in dose by more than 75%. When the waste nuclide mix is taken into account,  $\sim 97\%$  of the total dose commitment from an instantaneous release of disposed activity appears to be delivered by  $^{60}\text{Co}$  and  $^{137}\text{Cs}$ . The assumption of instantaneous nuclide release is, however, an extremely unrealistic one. In particular, the dissolution of  $^{60}\text{Co}$  from steel and other structural components is unlikely to be quantitative within the short mean lifetime ( $\sim 7.5$  y) of this nuclide. In addition, the model results show that the  $^{60}\text{Co}$  dose from any spontaneous release is delivered primarily ( $>90\%$ ) in the near-field disposal compartment, reflecting firstly its high degree of uptake on to suspended and bottom sediments and, again, its short half-life. Thus, from a very rapid release of reactor-derived radionuclides,  $^{137}\text{Cs}$  is indeed the main deliverer of dose beyond the immediate disposal region. For a slower release of radionuclides, e.g. after 500 years of containment, 99% of the dose commitment (3 man Sv) will arise from  $^{14}\text{C}$  and the remainder from  $^{59}\text{Ni}$  and  $^{63}\text{Ni}$ . The MEL ARCTIC2 results on  $^{60}\text{Co}$ , however, confirm its potential importance in subsequent local assessments.

### Conclusions

Both the preliminary radiochemical and modelling contributions by IAEA-MEL to the international consequence assessment programme suggest that, at least beyond the immediate vicinity of the dump-sites, the anthropogenic radioactivity enhancements resulting from the past disposals in the Arctic Seas are, and will continue to be, low and will result in doses which are comparable to or less than those resulting from other artificial and natural sources of radioactivity. The focus of future work should clearly lie on monitoring and modelling the dump-sites themselves, their environments and their local transfers and consequent exposures.

### Acknowledgements

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FIGURE 1

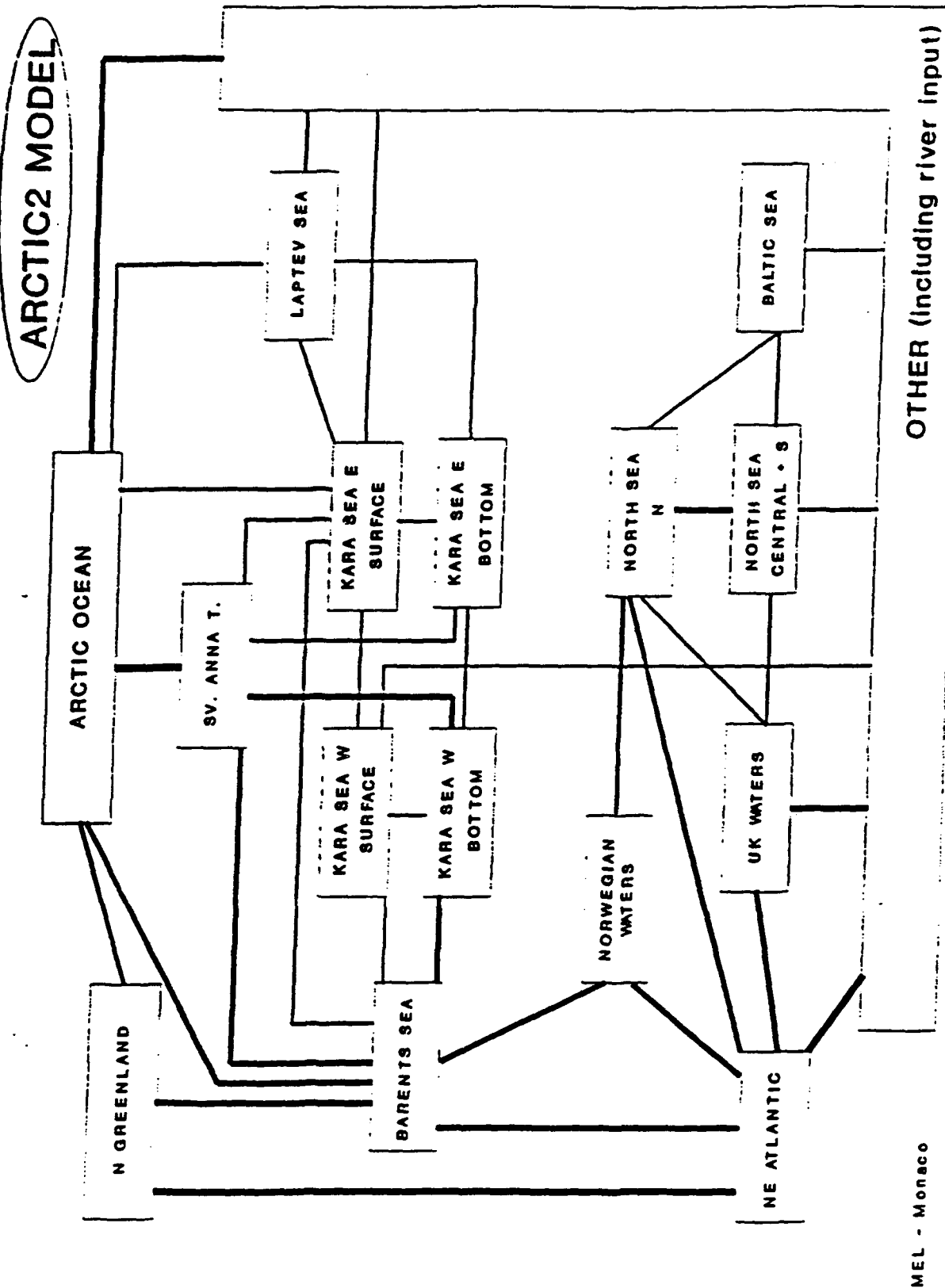


FIGURE 2

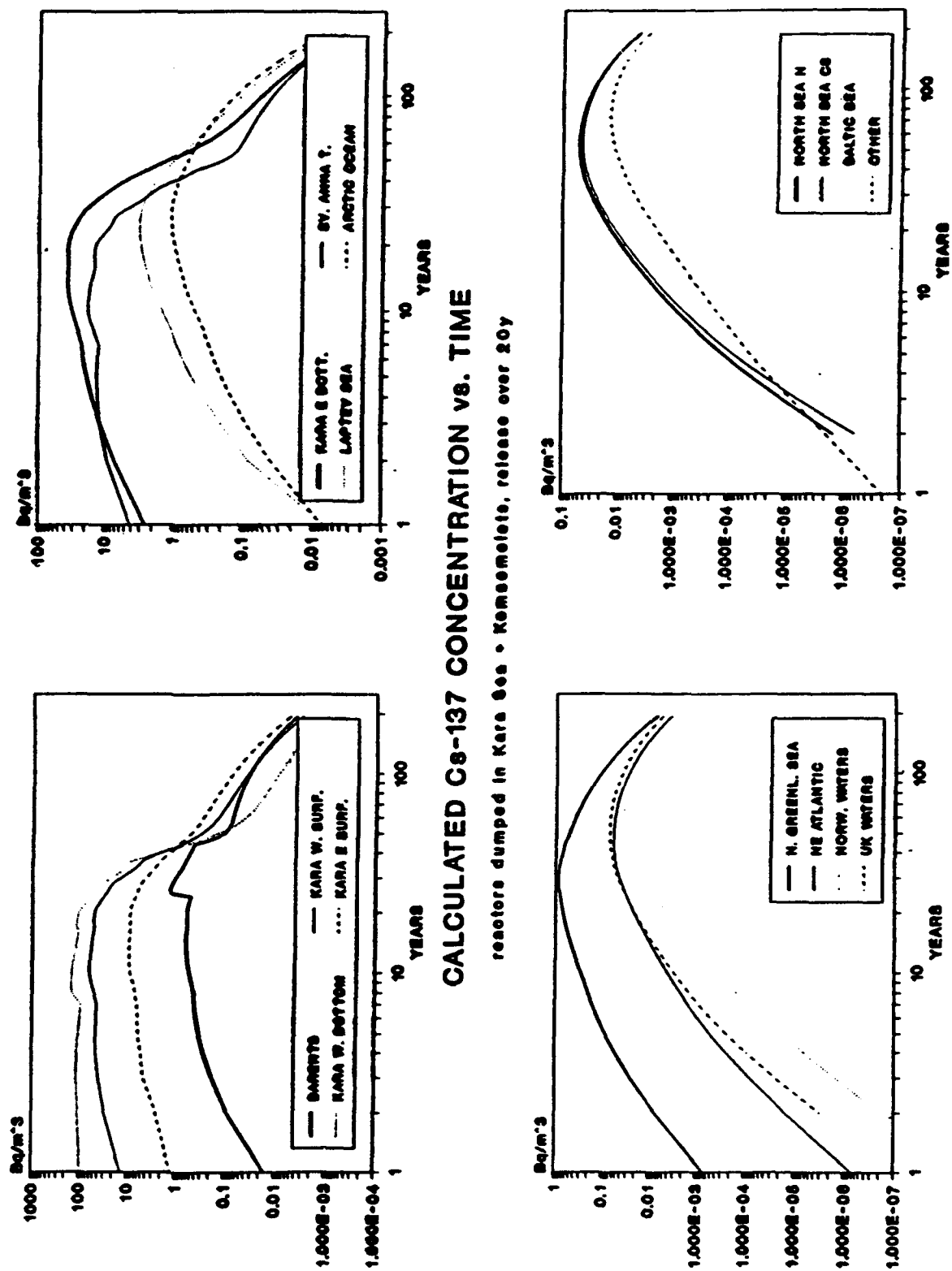


TABLE 1.

$^{137}\text{Cs}$  DOSE PREDICTIONS (WHITE BOOK 3)  
(FISH CONSUMPTION)

SOURCE/ASSUMPTIONS	DOSE COMMITMENT (MAN SV)	MAX. INDIVIDUAL DOSE ( $\mu\text{SV Y}^{-1}$ )
1. Liq. + solid non-reactor waste. 50% $^{137}\text{Cs}$ . instantaneous release at dumping	3.3	1
2. 7 reactors (86 PBq in Kara Sea). 50% $^{137}\text{Cs}$ + Komsomolets (2PBq)		
a) instantaneous release at dumping	145	58
b) release over 20y (quadratic) starting at dumping	120	15
c) release over 100y (quadratic) starting at dumping	72	9
d) instantaneous release 500y after dumping	$1.5 \times 10^{-3}$	$1.2 \times 10^{-3}$
e) no fish catch in Arctic Seas. instantaneous release at dumping	127	-

TABLE 2.

## DOSE COMMITMENT CONTRIBUTIONS

Long-lived nuclides	at the time of dumping		500y after dumping	
	activity (TBq)	dose (man Sv)	activity (TBq)	dose (man Sv)
Fission products	51700	37	6	0.002
Actinides	4100	0.09	83	0.07
Activation products	41700	176	130	3
	Total	213		3

assuming

- a) LLNL max. inventory
- b) no nuclide removal (ie max. dose)
- c) 300y truncation
- d) adult dose factors
- e) fish ingestion pathway





**Application of U.S. Interagency Coastal and Marine Monitoring Programs for Determining Levels of Radioactivity Resulting from Disposal of Radioactive Wastes in Arctic Seas**

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WHOI Conference on "Radioactivity and Environmental Security in the Oceans: New Research and Policy Priorities in the Arctic and North Atlantic," Woods Hole, MA, June 7-9, 1993

Introduction

Since World War II, the increased use of radioactive materials by many nations, has created substantial volumes of radioactive waste and byproduct materials. Today, problems associated with monitoring, storing, and disposal of these wastes and materials are of major concern. Disposal of radioactive waste/nuclear materials in coastal and deep ocean waters has been a key issue for scientists, environmentalists, and policy makers for many years. This conference, organized to evaluate current and future impacts of artificial radioactivity in the marine environment, reflects renewed worldwide concerns about potential environmental and human health effects resulting from disposal of radioactive waste materials in the Arctic seas. With those concerns in mind, this paper is a brief description of the Environmental Protection Agency's Office of Radiation and Indoor Air (EPA/ORIA) monitoring at the U.S. sites that were previously-used for ocean disposal of LLW, and also provides information about two cooperative U.S. interagency monitoring programs that are directly applicable to determining and assessing potential existing and/or future effects from nuclear contamination in the Arctic region. The two interagency programs include: (a) National Status and Trends (NS&T) monitoring for radioactivity in sediment and biota samples, conducted between 1986 and 1990 by the National Oceanic and Atmospheric Administration (NOAA) and the Environmental Protection Agency's Office of Radiation Programs (EPA/ORP); and, (b) Marketplace Seafood monitoring for radioactivity in seafoods, conducted in 1981 and 1982 by the Food and Drug Administration (FDA) and EPA/ORP.

Monitoring of Major U.S. Ocean Sites Used for Disposal of LLW

In 1974 the EPA/ORP (now EPA/ORIA), began to monitor ocean sites that had previously been used by the U.S. for disposal of LLW. Between 1974 and 1984, the four AEC-designated ocean sites that received the majority of LLW were surveyed and sampled from submersibles and surface ships. The four sites actually include five disposal locations, as follows: (a) two sub-sites in the Pacific Ocean, near the Farallon Islands (known as the 900m and 1700m sites, respectively); (b) two sites off the mid-Atlantic coast (known as the 2800m and 3800m sites, respectively); and, (c) Massachusetts Bay. Figure 1 shows the locations of the four major U.S. LLW disposal sites.

Monitoring included: (a) evaluating the condition of LLW containers after their descent to the seafloor and prolonged exposure to the environment; (b) identifying in-situ parameters and determining their potential to affect radionuclide transport or migrations; (c) assessing whether the effects from previous ocean disposals of LLW had adversely impacted human health or the marine environment, or whether any potential existed for future impact; and, (d) obtaining data to develop site designation, packaging and monitoring criteria to regulate any future ocean disposal of LLW.

The EPA/ORIA completed its studies at these sites in the 1980s and, since then, has continued to expand its database for radioactivity in the marine environment by working with NOAA to obtain samples for radionuclide analysis from the NOAA NS&T Monitoring Program, which was collecting samples from U.S. coastal areas that were either close to or between ocean sites previously used for disposal of LLW.

#### The National Status and Trends Monitoring Program

NOAA has primary responsibility for monitoring effects from disposal of materials into U.S. coastal and ocean waters, and has developed and implemented a series of monitoring programs over the years to assess and evaluate the quality of marine and estuarine environments. A new, broad-based program was implemented in 1984 to acquire reliable and continuous status and trends information that could be used to make decisions on the allocation and use of resources to protect the nation's coastal and estuarine environments. The specific objectives of this new NS&T monitoring program included defining the geographic distribution of contaminant concentrations in biological tissues and in sediments, determining temporal changes in those concentrations, and documenting biological responses to contamination. Since toxic chemicals pose some of the greatest threats to coastal and estuarine environments, the emphasis of the NS&T Program is on the measurement of these chemicals and observations of the effects they may cause. The NS&T Program is comprised of two monitoring components, Benthic Surveillance and Mussel Watch, to measure existing levels of toxic chemical contaminants in bivalves (mussels and oysters), benthic fish (flounder and other bottom-dwelling fish), and associated sediments [NOAA, 1988].

The Benthic Surveillance component measures existing levels of chemical contamination in seafloor sediments and bottom-feeding fish at key sites in the nation's estuaries and nearshore zone to determine the incidence of any diseases in benthic fish species. Sediment samples and benthic fish are analyzed for major and trace chemical elements and groups of organic compounds. Benthic fish are also examined for physical defects and histopathological effects. Target benthic fish species are demersal fish that consume benthic and epibenthic invertebrates and are widely distributed in the proposed sampling locations [NOAA, 1986].

Bivalve molluscs, which are reliable indicator organisms for

monitoring environmental contamination, are collected and analyzed in the Mussel Watch component. The blue mussel (*Mytilus edulis*) is collected along the northeast Atlantic coast. The American oyster (*Crassostrea virginica*) is collected from Delaware Bay on the Atlantic coast to lower Laguna Madre along the coast of Texas in the Gulf of Mexico. The California mussel (*Mytilus californianus*) and blue mussel are collected along the Pacific coast, including Alaska [NOAA, 1987] [Shigenaka and Lauenstein, 1988].

Both NS&T components are divided into four large areas (Atlantic coast, Gulf of Mexico, mainland Pacific coast, and Alaska) with about 75 Benthic Surveillance and 200 Mussel Watch sampling stations [Robertson, 1993]. During any given year, NOAA routinely collects sediment, benthic fish, and bivalves from approximately one-half of all the Benthic Surveillance and Mussel Watch monitoring locations.

#### NS&T Samples Collected for EPA

NOAA's original NS&T monitoring plans did not include collecting and analyzing NS&T samples for radioactivity. EPA/ORIA saw this as an opportunity to cost effectively expand its radioanalytical data base and, in 1986, requested samples from the Benthic Surveillance and Mussel Watch components for radionuclide analyses. NOAA began collecting samples for EPA in November 1986. Subsequent samples were collected in 1987, 1988, 1989 and 1990.

The overall EPA/ORIA objective for obtaining NS&T samples was to determine the levels of radioactivity, primarily for anthropogenic (man-made) radionuclides in the coastal marine environment. Specific applications of such data would support EPA responsibilities under the Marine, Protection, Research, and Sanctuaries Act of 1972. Another useful application would be to expand the Agency's Environmental Radiation Ambient Monitoring System (ERAMS) into the coastal marine environment, thereby establishing a baseline and trend assessment data base that could be used to detect any accidental releases of radioactivity. Thus, samples were requested and collected (Figure 2) from NS&T sites near previously used LLW disposal sites, or from areas between LLW sites where the EPA had not previously monitored for radioactivity.

Sediment samples were collected for EPA in 1986 and 1987 from Boston Harbor, Delaware Bay and Chesapeake Bay off the Atlantic coast, and from San Francisco Bay and the Farallon Islands off the Pacific coast. In 1988, east coast samples were collected from Boston Harbor and Raritan Bay (New Jersey), and from Coos Bay (Oregon) and San Francisco Bay off the west coast. Samples were collected in 1989 from Charleston Harbor (South Carolina) and Sapelo Sound (Georgia). In 1990, samples were again collected from Boston Harbor and San Francisco Bay. Benthic fish samples were collected from Boston Harbor, Delaware Bay, Chesapeake Bay and San Francisco Bay in 1986 and 1987. Attempts to acquire fish near the Farallon Islands disposal site in 1987 were not successful, but samples of the holothurian *Stichopus* sp. were obtained. The 1988

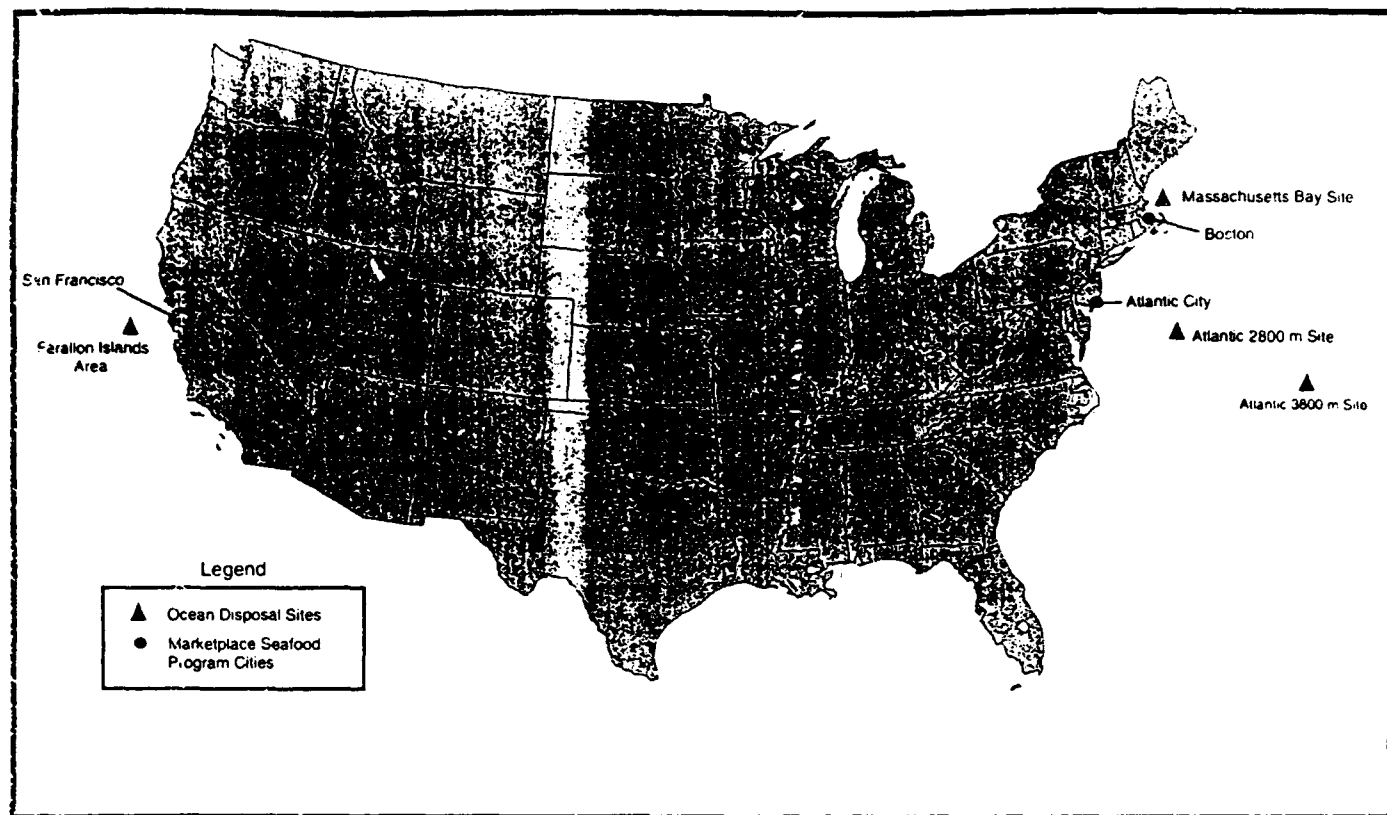


FIGURE 1. Locations of Marketplace Seafood Program Cities and Major U.S. Ocean Disposal Sites

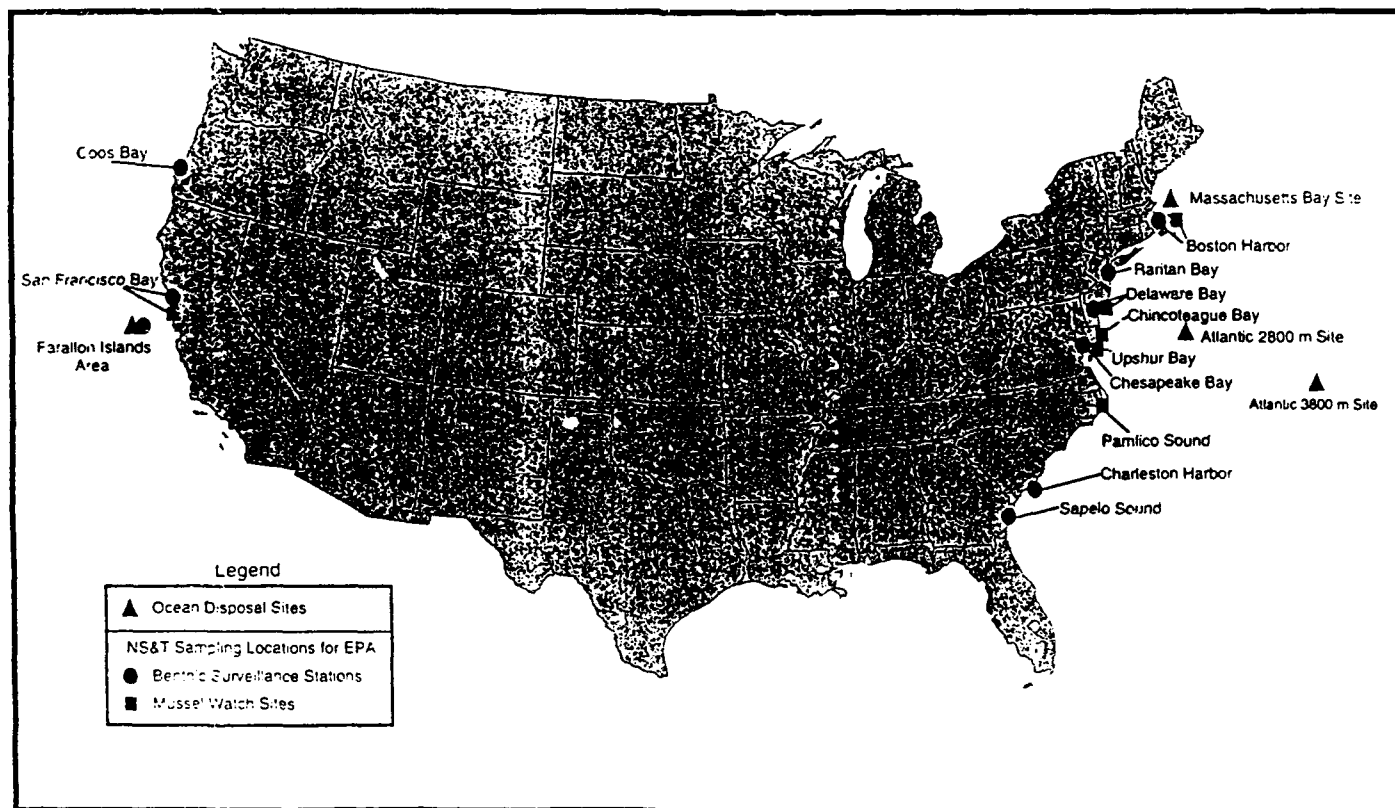


FIGURE 2. Locations of Major U.S. Ocean Disposal Areas for LLW, 1946-1977

samples were from west coast stations in Coos Bay (Oregon) and San Francisco Bay. In 1990, samples were collected from Boston Harbor and San Francisco Bay. Bivalves were collected from Deer Island and Brewster Island in Boston Harbor, False Egg Island Point (New Jersey) in Delaware Bay, Chincoteague Inlet (Virginia) in Chincoteague Bay, Quinby Inlet (Virginia) in Upshur Bay, and Wysoching Bay (North Carolina) in Pamlico Sound in 1986 and 1987; samples from the Pacific coast came from Dunbarton Bridge and San Mateo Bridge in San Francisco Bay. No samples were collected for EPA in 1988, 1989 or 1990.

Sediments were normally collected by box corers to minimize chances for surface disruption of the samples. Depending upon the size and condition of the box core samples, two or three subcore samples were extruded from each box core sample. Plastic subcore liners were used to extrude and contain smaller samples for post-survey radionuclide analysis. Subcore samples were usually no less than 10 to 15 cm long. Benthic fish samples were obtained by otter trawls or similar equipment. Upon collection, fish were sorted by species and then dissected. Samples preserved for radionuclide analysis were comprised of muscle (flesh), liver and stomach body parts from no less than ten specimens of each target fish species. Bivalves (mussels, oysters, clams) were collected by dredge, tongs, sampling fork, or by hand (depending on where the organisms were located and the water depth). Samples were then separated by type and sorted by size. Mussels with body length less than 5 cm or greater than 8 cm were discarded to avoid variability related to age and uptake. Oysters were discarded if they were less than 7 cm or greater than 10 cm long. Clams were collected when mussels and oysters were not available. The soft tissues from 10 or more bivalves collected at each sampling location were removed from the shells for radionuclide analysis [Lauenstein et al, 1987].

#### Radioanalytical Procedures

Sediment, benthic fish and bivalve samples collected during 1986-1987 and 1989-1990 were analyzed by the EPA/ORIA National Air and Radiation Laboratory (NAREL) in Montgomery, Alabama. Sediment and benthic fish samples collected in 1988 were analyzed by the TMA/Eberline laboratory in Albuquerque, New Mexico. Samples were prepared for analysis and analyzed according to the standard procedures at each laboratory [EPA/EERF, 1988] [HASL-300, 1990]. All samples were analyzed by gamma spectroscopy for cesium-137, and radionuclide-specific alpha spectroscopy for plutonium-238 and plutonium-239, 240. Sediment subcores were divided into 0-5, 5-10, and 10-15 cm increments and freeze dried prior to their analysis. If sufficient benthic fish and bivalve tissue samples were available after the plutonium analyses were completed, strontium-90 analyses were performed. Counting times for both gamma and alpha spectroscopy were normally 3000 minutes for each sample. In reporting data numerical results were corrected for radiodecay to the sampling date. Counting errors are reported at the 2-sigma (95 %) confidence level. Analyses with counting errors (2-sigma) greater than 50 % are reported as not detected (ND). Radionuclide

concentrations are in picocuries per gram (pCi/g) dry weight.

### Analytical Results

Summaries of data are given in Tables 1 and 2 for sediment samples, and are organized by sampling area and year of collection. Mean activity is presented for each 5-cm increment of subcores where good vertical stratification was obtained, and for whole samples when bottom grabs were collected. Ranges of activity include analytical results for all samples yielding detectable results within a particular subcore increment, or where more than one bottom grab was collected at a sampling location. Where mean activity is the only result reported, with no accompanying range data, only one of the subcores collected at a sampling station yielded a measurable activity for that particular radionuclide. Radionuclide activity concentrations in samples with 2-sigma counting errors greater than 50 percent are reported as not-detected. Thus, because the counting errors approximated 60 percent, all analyses for plutonium-238 are reported as ND (not detected). Cesium-137 was detected in sediment samples from all sites except Chesapeake Bay; plutonium-239, 240 was detected in all of the sites. Plutonium-238 was not detected in any of the Atlantic or Pacific coast samples. The mean activity for cesium-137 ranged from 0.02 to 0.12 pCi/g (picocuries per gram) in Atlantic coast samples, and from 0.02 to 0.16 in Pacific coast samples. For plutonium-239, 240, mean activity ranged from 0.01 to 0.04 pCi/g in Atlantic samples, and from 0.01 to 0.02 in Pacific samples. These data conform to activity reported in historical data that are attributed to fallout from nuclear weapons testing [IAEA, 1976] [Noshkin, 1978] [Livingston and Bowen, 1979]. The data also fall within the ranges of activity reported from previous EPA/ORIA surveys at U.S. ocean sites used for disposal of LLW [Dyer, 1976] [Curtis and Mardis, 1984].

The summaries of data for biota (benthic fish and bivalves) in Tables 3, 4, and 5 show essentially non-detectable results. Cesium-137 was detected only in the muscle of Winter Flounders collected from Boston Harbor in 1987, and in the stomach of Starry Flounders collected from Coos Bay in 1988. Mean activities for these samples were 0.04 and 0.03 pCi/g, respectively. Cesium-137 was not detected in any of the bivalve samples. Plutonium-239, 240 was detected in Windowpane Flounder stomach samples collected from Delaware Bay in 1987 (0.01 pCi/g), and in holothurians collected near the Farallon Islands site in 1987 (0.02 pCi/g). Plutonium-239, 240 was also detected in bivalve samples from Delaware and Upshur Bays (0.02 and 0.01 pCi/g respectively). Plutonium-238 was not detected in any of the benthic fish or bivalve samples. These data are within the ranges reported in historical data as being attributable to fallout from nuclear weapons testing [IAEA, 1976] [Curtis and Mardis, 1984] [Curtis, 1988]. Strontium-90 was not detected in any of the benthic fish composite body parts (livers, muscles, stomachs) from Atlantic or Pacific coast sampling stations. However, no samples containing bone material were analyzed for strontium. Strontium-90 was also not detected in any

TABLE 1. Summary, Radionuclide Activity in Atlantic Coast Sediment Samples

Sampling Area	Year	Subcore Segment (cm)	Cs-137 Activity Mean (Range)	Pu-238 Activity Mean (Range)	Pu-238, 240 Activity Mean (Range)
Boston Harbor	1987	0-5	0.06 (0.005 - 0.096)	ND	0.02 (0.011 - 0.032)
		5-10	0.03 (0.010 - 0.051)	ND	0.03 (0.010 - 0.048)
		10-15	0.05	ND	0.03
	1988	Grab	0.02	ND	0.01
	1989	0-5	0.09 (0.05 - 0.13)	ND	0.02 (0.014 - 0.043)
		5-10	0.06 (0.05 - 0.13)	ND	0.03 (0.010 - 0.044)
		10-15	0.12 (0.10 - 0.17)	ND	0.04 (0.019 - 0.051)
Portau Bay	1988	Grab	0.04	ND	0.02
Delaware Bay	1987	0-5	0.05 (0.034 - 0.071)	ND	0.01 (0.007 - 0.016)
		5-10	0.03 (0.020 - 0.038)	ND	0.03 (0.007 - 0.061)
		10-15	0.04 (0.020 - 0.072)	ND	0.01 (0.000 - 0.014)
Chesapeake Bay	1987	0-5	ND	ND	0.01 (0.004 - 0.012)
		5-10	ND	ND	0.01 (0.001 - 0.009)
		10-15	ND	ND	0.01 (0.000 - 0.016)
Charleston Harbor	1988	Grab	0.09	ND	0.02
Sapelo Sound	1988	Grab	0.05	ND	0.01

## TABLE Notes:

Activity units are pCi/g dry.

Approximate 2-sigma counting errors are: 35% for cesium-137 and 20% for plutonium-239/240.

ND = Not Detected

TABLE 2. Summary, Radionuclide Activity in Pacific Coast Sediment Samples

Sampling Area	Year	Subcore Segment (cm)	Cs-137 Activity Mean (Range)	Pu-238 Activity Mean (Range)	Pu-238, 240 Activity Mean (Range)
San Francisco Bay	1987	0-5	0.02 (0.012 - 0.034)	ND	0.01 (0.000 - 0.012)
		5-10	0.02 (0.010 - 0.029)	ND	0.01 (0.000 - 0.014)
		10-15	0.02 (0.013 - 0.028)	ND	0.01 (0.000 - 0.009)
	1988	Grab #1	0.03	ND	0.01
		Grab #2	0.05	ND	0.01
	1989	0-5	0.16 (0.07 - 0.39)	ND	0.01 (0.003 - 0.007)
Paralau Islands	1987	5-10	0.14 (0.06 - 0.39)	ND	0.01 (0.002 - 0.014)
		5 Grabs	0.04 (0.034 - 0.055)	ND	0.02 (0.016 - 0.025)
Cape Bay	1988	Grab	0.07	ND	0.02

## TABLE Notes:

Activity units are pCi/g dry.

Approximate 2-sigma counting errors are: 35% for cesium-137 and 20% for plutonium-239/240

ND = Not Detected

TABLE 4. Summary, Radionuclide Activity in Benthic Fish Samples (Pacific Coast)

Sampling Area	Year	Fish Analyzed	Body Part	Cs-137 Activity (pCi/g dry)	2-Sigma Counting Error	Pu-238 Activity (pCi/g dry)	2-Sigma Counting Error	Pu-239, 240 Activity (pCi/g dry)	2-Sigma Counting Error
San Francisco Bay	1987	Starry Flounder	Liver	ND	---	ND	---	ND	---
			Muscle	ND	---	ND	---	ND	---
			Liver	ND	---	ND	---	ND	---
			Muscle	ND	---	ND	---	ND	---
Coos Bay	1988	White Croaker	Stomach	ND	---	ND	---	ND	---
			Liver	ND	---	ND	---	ND	---
			Muscle	ND	---	ND	---	ND	---
			Stomach	ND	---	ND	---	ND	---
Farallon Islands	1987	Sea Cucumber (*)	Liver	ND	---	ND	---	ND	---
			Muscle	ND	---	ND	---	ND	---
			Stomach	ND	---	ND	---	ND	---
			Whole Body	0.88	+/- 0.012	ND	---	0.08	+/- 0.004

## NOTES TO TABLE:

Body parts are composites from 10 or more specimens of the benthic fish collected.

(\*) = the holothurian (*Stichopus sp.*) was collected and analyzed instead of fish

ND = Not Detected

TABLE 5. Summary, Radionuclide Activity in Benthic Fish Samples (Atlantic Coast)

Sampling Area	Year	Fish Analyzed	Body Part	Cs-137 Activity (pCi/g dry)	2-Sigma Counting Error	Pu-238 Activity (pCi/g dry)	2-Sigma Counting Error	Pu-239, 240 Activity (pCi/g dry)	2-Sigma Counting Error
Delaware Bay	1987	Winter Flounder	Liver	ND	---	ND	---	ND	---
			Muscle	ND	---	ND	---	ND	---
			Stomach	ND	---	ND	---	ND	---
			Whole Body	0.8	+/- 0.05	ND	---	ND	---
Boston Harbor	1987	Winter Flounder	Liver	ND	---	ND	---	ND	---
			Muscle	ND	---	ND	---	ND	---
			Stomach	ND	---	ND	---	ND	---
			Whole Body	ND	---	ND	---	ND	---
Chesapeake Bay	1987	Atlantic Croaker	Liver	ND	---	ND	---	ND	---
			Muscle	ND	---	ND	---	ND	---
			Stomach	ND	---	ND	---	ND	---
			Whole Body	ND	---	ND	---	ND	---

## NOTES TO TABLE:

Body parts are composites from 10 or more specimens of the benthic fish collected.

ND = Not Detected

TABLE 6. Summary, Radionuclide Activity in Bivalve Samples

Sampling Location/Area	Cs-137 Activity (pCi/g dry)	2-Sigma Counting Error	Pu-238 Activity (pCi/g dry)	2-Sigma Counting Error	Pu-239, 240 Activity (pCi/g dry)	2-Sigma Counting Error
Boston Harbor Deer Island	ND	---	ND	---	ND	---
Boston Harbor Brewster Island	ND	---	ND	---	ND	---
False Egg Island Point (NJ) Delaware Bay	ND	---	ND	---	0.02	+/- 0.008
Chincoteague Inlet (VA) Chincoteague Bay	ND	---	ND	---	ND	---
Quincy Inlet (VA) Upshur Bay	ND	---	ND	---	0.01	+/- 0.008
Wyebooching Bay (NC) Pamlico Sound	ND	---	ND	---	ND	---
San Francisco Bay Dumbarton Bridge	ND	---	ND	---	ND	---
San Francisco Bay San Mateo Bridge	ND	---	ND	---	ND	---

## NOTES TO TABLE:

The sample from each area represents a composite of approximately 15 bivalves.

Analyses were conducted on soft tissue only; shells were not analyzed.

ND = Not Detected



of the composite bivalve soft tissue samples. Bivalve shells were not analyzed for strontium-90.

Individual radionuclide analysis results (not summarized data) for each individual sediment and biota sample collected during this study will be published in an EPA/ORIA report [Curtis, 1993].

#### Monitoring Program for Radionuclides in Marketplace Seafoods

In 1981 EPA/ORIA requested public health monitoring assistance from FDA to supplement EPA's monitoring of environmental effects at and near LLW disposal sites. Since contamination of seafoods is the only significant pathway by which humans could be affected by ocean disposals of LLW, this monitoring would be conducted to determine levels of radioactivity in seafoods commercially available from marketplaces in cities near major U.S. ocean sites previously used for disposal of LLW. The FDA had maintained a national Total Diet Compliance Program for many years to monitor levels of radioactivity in foods, and had also analyzed 12 fish samples, collected near the Farallon Islands LLW disposal site by commercial fishermen in 1978 and 1980, for radioactivity [Curtis, 1988]. From its own studies, and the results of EPA studies at major ocean disposal sites, the FDA had already concluded that previous U.S. ocean disposals of LLW were not giving rise to radiation levels in commercial fish above background levels of radioactivity in the oceans. The FDA did concur with EPA that the available data on radioactivity in commercial fish species were limited and that further sampling was warranted [Stroube, 1984]. The FDA thus agreed to initiate a limited Marketplace Seafood Sampling and Analysis Program for EPA/ORIA.

In 1981 and 1982, FDA district offices in California, Massachusetts and New Jersey collected samples of edible fish and eels for radionuclide analysis marketplace fishermen in San Francisco, Boston, and Atlantic City, which are the three major cities closest to the major U.S. ocean sites for disposal of LLW (Figure 1). Samples were obtained, when possible, from bottom-feeding fish species that were most likely be eaten by the public and caught by commercial fishermen near the major LLW disposal sites. Bottom-feeders were the species of choice since they would most likely show any evidence of radioactivity migrating into the environment from the LLW containers in the disposal sites. When bottom-feeders were not available, samples of nonmigratory (preferably) or migratory pelagic feeders, or shellfish were to be collected as the fishing boats docked to sell their catches at waterfront markets. Field personnel from each FDA district were instructed to obtain three different species of bottom-feeders and sufficient numbers of each species to obtain twenty pounds of edible portions of fish per species for radionuclide analysis.

Samples collected included: cod, haddock, pollock, hake, flounder, and cusk eels from Boston; tilefish and Conger eels from Atlantic City; and sole, lingcod, hake, red snapper, sablefish, and and thornyheads from San Francisco. All samples were sent to the

FDA's Winchester Engineering and Analytical Center (WEAC) in Massachusetts where a blended homogenate was prepared from the edible portions of the 20 pounds of fish sample from each species collected. One-half of each blended homogenate was retained for analysis by WEAC and the other half was forwarded to the EPA/ORIA's National Air and Radiation Environmental Laboratory (NAREL) in Montgomery, Alabama.

#### Radioanalytical Procedures

The FDA analyzed each blended homogenate sample for gross gamma activity, cesium-137, iodine-131, barium-140, and strontium-90. Six of the twelve or thirteen samples received from an FDA district were also analyzed for plutonium-239. Analytical methods used were well-established methods that have routinely been used by WEAC to analyze thousands of samples in the FDA Radionuclides in Foods Compliance Program [Baratta, 1969] [Baratta, 1977] [PHS, 1967]. The EPA analyzed each of its blended homogenate samples for gross gamma, alpha and beta activity. In addition, individual analyses were conducted for lead-210, polonium-210, strontium-90, and radium-226, as well as for thorium-227, -228, -230, and -232, for uranium-235 and -238, and for plutonium-238, and -239, -240. Methodologies for preparing samples for analysis, and analytical procedures are described in the NAREL Radiochemistry Procedures Manual [EPA/EERF, 1988].

#### Analytical Results

The data from EPA's analysis of 41 blended homogenate samples is shown in Tables 6, 7, and 8. Cesium-137 was detected in cusk eels collected from Boston, and in some of the Lingcod, Sablefish, and Thornyhead fish samples collected from San Francisco. No cesium-137 was detected in any of the Atlantic City samples. Plutonium-238 was detected in one Tilefish samples collected from Atlantic City. Plutonium-239, -240 and strontium-90 were not detected in any samples. The levels of cesium-137 and plutonium-238 activity detected are within the ranges of cesium-137 data detected in benthic fish collected and analyzed for radioactivity under the NS&T monitoring program, and also conform to the levels of activity reported in historical data that are attributed to fallout from nuclear weapons testing [IAEA, 1976] [Noshkin, 1978] [Livingston and Bowen, 1979]. The data also fall within the ranges of activity reported from previous EPA/ORIA surveys at U.S. ocean sites used for disposal of LLW [Dyer, 1976] [Curtis and Mardis, 1984]. The analytical data from the FDA analysis of blended homogenate samples was within the ranges reported from EPA analysis of blended homogenate samples [Curtis, 1988].

Both FDA and EPA analyzed the blended homogenate samples for natural and certain man-made (anthropogenic) radionuclides, but the data shown in the Tables 6-8 is restricted to activity levels for the man-made radionuclides cesium-137, plutonium-238, and plutonium-239, -240 in seafood samples. This is done for the sake of brevity: the purpose of the Marketplace Seafood study was to

TABLE 6. Radionuclide Activity in Marketplace Seafoods  
Collected From Boston Fishermen, Analyzed by EPA

Sample	Date Collected (mo/da/yr)	Sample Number	Cs-137 Activity (Error)	Pu-238 Activity (Error)	Pu-239, 240 Activity (Error)
Winter Flounder	09/02/81	15991	ND	ND	ND
		15992	ND	ND	ND
Atlantic Cod	09/02/81	15993	ND	ND	ND
		15994	ND	ND	ND
Red Hake	09/10/81	16842	ND	ND	ND
Cusk eel	09/10/81	16843	0.031 (+/- 0.011)	ND	ND
Winter Flounder	09/21/81	16844	ND	ND	ND
Atlantic Cod	09/21/81	16845	ND	ND	ND
Winter Flounder	09/10/82	26958	ND	ND	ND
Red Hake	09/13/82	26959	ND	ND	ND
Atlantic Cod	09/13/82	26960	ND	ND	ND
Haddock	11/02/82	26961	ND	ND	ND
Atlantic Cod	11/02/82	26962	ND	ND	ND
Pollock (Boston Blackfish)	11/02/82	26963	ND	ND	ND

TABLE NOTES:

Samples are blended homogenates of the edible portions of composite fish samples  
Radionuclide activity is reported in pCi/g wet weight  
ND = Not Detected (activity is either not present or the 2-sigma counting error > 50 %)

TABLE 7. Radionuclide Activity in Marketplace Seafoods  
Collected From Atlantic City Fishermen, Analyzed by EPA

Sample	Date Collected (mo/da/yr)	Sample Number	Cs-137 Activity (Error)	Pu-238 Activity (Error)	Pu-239, 240 Activity (Error)
Tilefish	09/03/81	16846	ND	ND	ND
Conger eel	09/13/81	16847	ND	ND	ND
Conger eel	09/24/81	17633	ND	ND	ND
Tilefish	09/24/81	17634	ND	ND	ND
Tilefish	09/30/81	17635	ND	ND	ND
Tilefish	10/07/81	17636	ND	ND	ND
Tilefish	04/03/82	23031	ND	ND	ND
Tilefish	04/03/82	23032	ND	ND	ND
Tilefish	06/02/82	23033	ND	ND	ND
Tilefish	05/07/82	23034	ND	ND	ND
Tilefish	05/10/82	23035	ND	ND	ND
Tilefish	05/10/82	23036	ND	0.0002 (+/- 0.0001)	ND

TABLE NOTES:

Samples are blended homogenates of the edible portions of composite fish samples  
Radionuclide activity is reported in pCi/g wet weight  
ND = Not Detected (activity is either not present or the 2-sigma counting error > 50 %)

TABLE 8. Radionuclide Activity in Marketplace Seafoods  
Collected From San Francisco Fishermen, Analyzed by EPA

Sample	Date Collected (mo/da/yr)	Sample Number	Cs-137 Activity (Error)	Pu-238 Activity (Error)	Pu-239, 240 Activity (Error)
Dover Sole	10/06/81	17639	ND	ND	ND
Thornyhead (Hard Head)	10/07/81	17637	ND	ND	ND
Pacific Hake (Butterfish)	10/07/81	17638	ND	ND	ND
Sablefish (Black Cod)	10/16/81	17640	ND	ND	ND
		17640X	ND	ND	ND
Lingcod	11/12/81	18620	0.007 (+/- 0.002)	ND	ND
		18620X	ND	ND	ND
Sablefish (Black Cod)	11/12/81	18621	0.007 (+/- 0.002)	ND	ND
Thornyhead (Hard Head)	11/12/81	18622	0.012 (+/- 0.002)	ND	ND
Pacific Red Snapper	05/26/82	23041	ND	ND	ND
Lingcod	05/26/82	23042	ND	ND	ND
Lingcod	05/27/82	23037	ND	ND	ND
Sablefish (Black Cod)	05/27/82	23038	ND	ND	ND
Pacific Red Snapper	05/27/82	23039	ND	ND	ND
Sablefish (Black Cod)	05/27/82	23040	ND	ND	ND

TABLE NOTES:

Samples are blended homogenates of the edible portions of composite fish samples  
Radionuclide activity is reported in pCi/g wet weight  
ND = Not Detected (activity is either not present or the 2-sigma counting error > 50 %)

determine potential effects from ocean disposal of man-made radionuclides that are associated with ocean disposal of LLW; and, the concentrations of these specific radionuclides are reported in the data published about disposals of nuclear materials by the Soviet Union in the Arctic region. Strontium-90 data is not included in the Tables because there were no detectable levels of activity reported by either laboratory from analysis of any of the Marketplace Seafood samples.

The data from the Marketplace Seafood Program clearly showed that the levels of man-made radionuclides in the collected seafoods were at or below those levels normally found in FDA-monitored foods [Stroube, 1984]. Detectable levels of cesium-137 and plutonium-238 activity were only found in 16 of the 239 analyses (6.7 percent) completed by the FDA and EPA laboratories. All analyses for plutonium-239, -240 and strontium-90 resulted in less than minimum detectable activity levels. All results were found to be in the lower portion of Range 1 of the Federal Radiation Council's guidelines for radioactivity, and were also found to be similar to previous FDA data from the analysis of imported fish obtained from retail outlets in the United States [Jelinek, 1982].

#### Application of NS&T-type and Marketplace Seafood-type Monitoring to Arctic Contamination

The data presented in this paper clearly show no significant radiation contamination. What is significant and applicable to the Arctic contamination problem is that the monitoring programs in place to effectively assess potential effects to Alaskan and other coastal areas from the disposal of nuclear materials by the former Soviet Union into marine environments. NOAA has already established six Alaskan NS&T monitoring stations. EPA and NOAA scientists and managers who were directly involved in the NS&T monitoring described in this paper have informally discussed adding more NS&T monitoring stations along the north and west coasts of Alaska. Figure 3 shows the existing NS&T monitoring stations, as well as locations suggested by EPA/ORIA to supplement the existing NOAA stations, which in toto could serve as a marine environmental "dew line" system. This two-fold increase in NS&T monitoring stations in Alaskan coastal areas could be a more effective tool for identifying increased levels of radioactivity in Alaskan coastal sediments and biota, that may be due to radionuclide transport from nuclear waste disposal sites. Technical and logistical/cooperative details need to be worked out and formalized between EPA, NOAA, and other interested/affected parties, but it appears that there is positive interest and a definite need to acquire baseline and trend assessment radionuclide data in this area.

The NS&T type of monitoring system could also be rather easily supplemented by a Marketplace Seafood type of program to identify whether radionuclides from the Arctic region disposals are being transported to humans via seafood pathways. Samples could be selected from localized and regional population centers in and

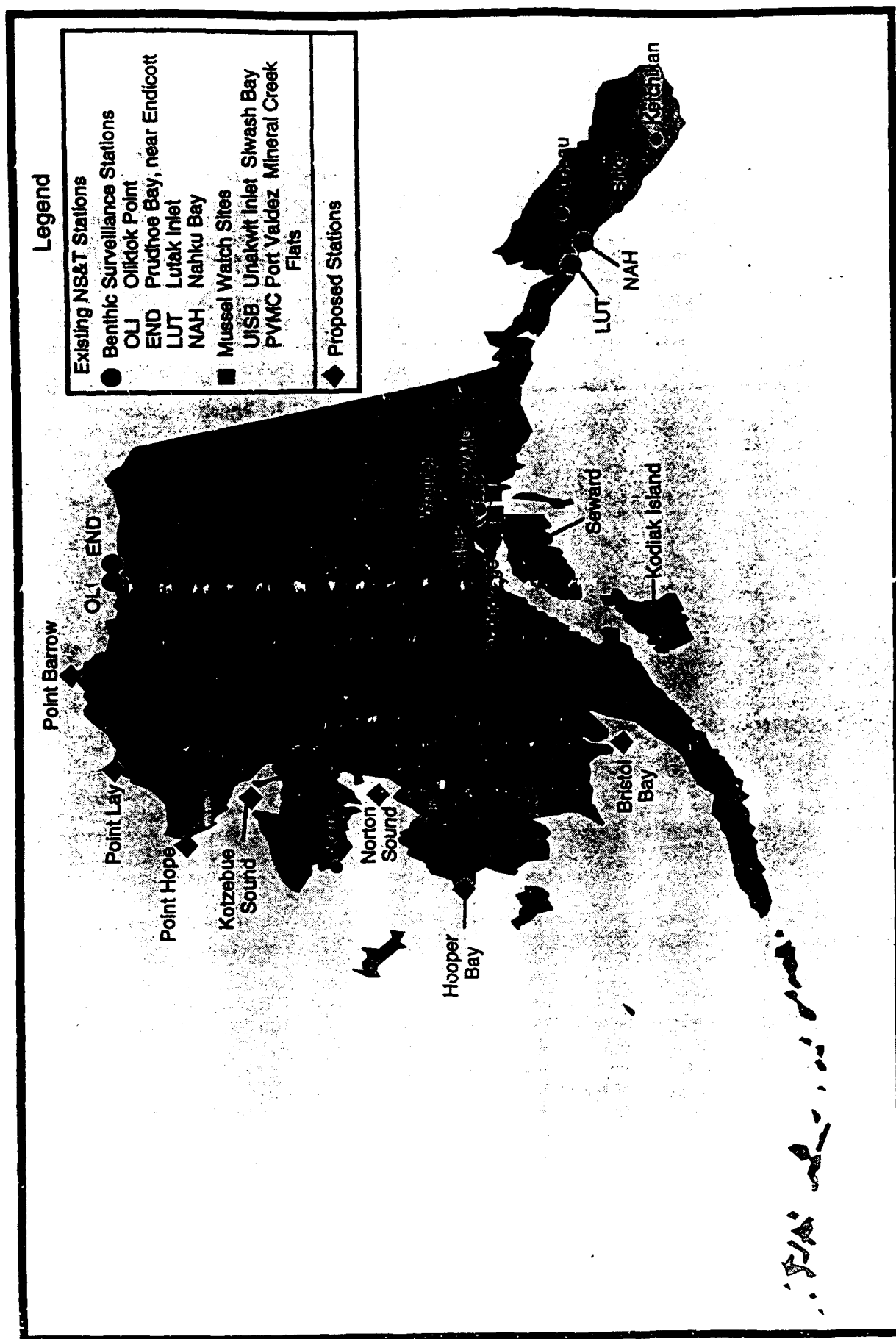


FIGURE 3. NS&amp;T Site Location Map - Alaska

across the Arctic region (i.e., in Alaska, Canada, Russia, and the Scandinavian countries) to identify and assess the levels of radioactivity in seafoods, and even in marine mammals, consumed by those populations. NS&T and Marketplace Seafood types of programs are effective tools for assessing and evaluating marine environmental quality. The NS&T Program is already in place and successfully operating. A marketplace-type program can be implemented rather easily and can be just as successful. Both can provide baseline and trend assessment data that can be utilized for many environmental effects studies. The EPA/ORIA, by working together with NOAA and the FDA, has broadened its capabilities to identify increased levels of radioactivity in coastal and marine environments. Such data, whether baseline or trend assessment in nature, can be applied to studying effects from any future nuclear power accidents that occur, or to such problems as the one being addressed by this conference. The tools are available to identify and assess radioactive contamination from the Arctic.

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# TREATMENT OF DATA ON SEA POLLUTIONS AND DUMPING SITES WITH RADIOECOLOGICAL INDICES METHOD

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## ABSTRACT

The concept of an "ecological meter (EM)" (i.e. of a system of globally ordered and ranked multidimensional images) used to fold all available radio-ecological information on sea dumping sites into system indexes diagnosing ecological peculiarities of sea ecosystems and environment pollution process is presented. The EM concept is realized through interactive procedures which represent composition of methods of factor, cluster, discriminant analysis, non-linear scaling on the basis of the Monte Carlo method. It is proposed in this paper to organize an international project aimed toward creation of EMs for ranking two types of radioecological risks: one corresponding to local and the other to global considerations of sea dumping sites. Such EMs would provide the possibility to watch over evolution of ecological state of environment, especially hydrobionts, optimize further monitoring and predict dangerous situations.

## I. METHODOLOGY

A lot of examples of system analysis of radioecological data can be submitted.

So, global dropdowns are analyzed on the basis of  $^{137}\text{Cs}/^{90}\text{Sr}$  ratio, geological age is identified by  $^{14}\text{C}/^{12}\text{C}$  ratio, lag of radionuclei in a cow is estimated by  $^{89}\text{Sr}/^{90}\text{Sr}$  ratio, etc. An impressive example of effectiveness of such methods of system analysis is discovery of the natural nuclear reactor in South Africa, based on  $\text{U}^{235}/\text{U}^{238}$  ratio in local uranium mines. However, correlation between only two radionuclei is commonly used. Here we shall consider methods of system analysis of radioecological monitoring of hydroenvironmental data for multidimensional case in terms of "ecological meter (EM)" concept, i.e. globally ordered and ranked system of multidimensional images<sup>1,2</sup>.

The concept supposes folding of large arrays multidimensional and noised radioecological data (radionuclei spectrum in trofic chains in sea ecosystems, in water, in silt sediments and hydrobiological, hydrochemical and hydrophysical parameters into generalized indexes and their reflecting onto specially constructed scales. Units on these scales are stated by reflecting on them some radioecological states which could be determined as reference points (standard, expert, modeled). Fig. 1 symbolically illustrates stated above: in some multidimensional metrical space trajectory  $T$  is constructed, states  $a_i$  being projected on it (thus they are ranked globally); some of them are declared as reference points. The lower part of the picture shows a typical structure of radioecological monitoring data: parameters  $x_i$  carry no information by themselves (observation data about

<sup>1</sup> V. B. Georgievsky, I.P. Kameneva. Interactive data analysis in ecological monitoring problems. // Problems of energy economy, 1991, vol. 1 pp 1-10.

<sup>2</sup> V. B. Georgievsky, I.P. Kameneva, A.P. Survila. Multidimensional analysis of ecological monitoring data. // Institute of Energetics Modeling Problems of Ukraine Academy of Sciences, Pub. No 92-45.

state  $a_i$  against  $x_i$  and  $x_{i+1}$  parameters override observation data about state  $a_2$ ); thus radioecological state can be diagnosed only by  $x_i$  and  $x_{i+1}$  complex.

Analysis of radioecological monitoring data is brought about studying evolution of points  $a_T$  along  $T$  trajectory in terms of distances between the reference points  $a_{K_T}$ ; location of points  $a_{K_T}$  represents radioecological state of object  $a_i$ . Prognoses of the state  $a_K$  and its rating are carried out in terms of the distance  $L$  along the scale  $T$ . Analysis of informative capacity of  $x_i$  indexes (optimization of observation regulation and estimation of efficiency of state management) is obtained analyzing sensitivity  $\partial a_T / \partial x_i$ .

The methodology of analysis of radioecological data is based on suppose that in multidimensional space formed by  $x_i$  there are compact structures  $a_i$  characterizing deep inner mechanisms of radioecological processes, caused by given sea waste disposal.

## **II. EM REALIZATION**

The structure of monitoring data  $a_i$  being studied is processed interactively by special hierarchical multilevel procedures, the latter being a composition of algorithms of component, cluster and factor analysis, non-linear scaling, building of state trees, sensitivity analysis. All procedures of data structure  $a_i$  study are carried out in frames of statistical experiments (Monte Carlo methods). Each set of algorithms is tested on specially simulated statistical data having preliminary known structure. All really observed data of radioecological monitoring  $x_i$  are studied by laying noises over them.

Criteria for constructing all mentioned above analysis procedures is experimental detection of maximal possible discrimination (in terms of statistics) of radioecological states. The following condition is imposed: one must provide non-deterioration of state  $a_i$  discrimination while widening list of ecological parameters  $x_i$  due to addition of intense positively correlated attributes. (Such algorithms of multidimensional analysis are suitable for handling all available information, particularly that requiring introduction Mahalanobis metrics.) Such procedures make it possible to analyze data having 100-parameters dimension. Analysis of each of these parameters  $x_i$  uses statistics of 100÷1000 tests.

## **III. ILLUSTRATIVE SAMPLES**

### **A.**

Radioecological monitoring data concerning silt sediments from the bottom of the cooling pool of Chernobyl nuclear power plant (NPP) was analyzed upon 7 radionuclides:  $^{110m}\text{Ag}$ ,  $^{134}\text{Cs}$ ,  $^{137}\text{Cs}$ ,  $^{89}\text{Sr}$ ,  $^{90}\text{Sr}$ ,  $^{106}\text{Ru}$  in 80 different locations  $St_1\{^{110m}\text{Ag}, \dots\}$ ,  $St_2\{^{110m}\text{Ag}, \dots\}$ , ...,  $St_{80}\{^{110m}\text{Ag}, \dots\}$  of aquatory<sup>3</sup>. Peculiarity of the structure of these data was in great difference (some orders) in radionuclides concentration and their relation from one point to another.

As the result of analysis it was revealed that radioecological state characterized by vector of 7 radionuclides  $St_i\{^{110m}\text{Ag}, \dots\}$  in all points of aquatory but for "A" point can be ordered along some straight line (Fig. 2) having accumulated dispersion about 85%. The state of one point — location "A" of aquatory was essentially different from other states, ordered along that line, though values of radionuclides concentrations in it didn't

<sup>3</sup>Nedostup L.M., Atomproject, Russia. Private conversation.

exceed concentrations measured in other points of the pool. It turned out that a lot of radionuclei were buried in that location.

#### B.

It was stated above that analysis of real radioecological data in terms of EM concept includes analysis of data by specially simulated numerical experiments.

One simple example of this statement is shown on Fig. 3. The subject of analysis was evolution of radionuclei mixture from RBMK-1000-type reactor of the Chernobyl NPP. Information on the mixture composition contained some additional irrelevant data. It turned out that radionuclei mixture from the reactor evolved along  $T$  curve having accumulated dispersion about 100%, irrelevant data being filtered away — points a, b, c on Fig. 3.

#### C.

The EM concept has been tested and practically used analyzing hydrobiological and hydrochemical regimes of Volga and Dnepr hydrosystems, Ladoga lake, lakes of Lithuania, data of remote space satellite monitoring registered by the United Net of Observation and Control of the USSR State Committee on Meteorology (1980-1990). It proved to be that all information collected during 20-years period of observations can be reflected almost without any losses onto one- and two-dimensional scales (Fig. 4).

### IV. POSSIBILITY OF CREATION OF "RADIOECOLOGICAL METERS" FOR SEA DUMPING SITES

Here we shall state some proposals, around which efforts of a number of organizations and explorers could be joined in order to: first, give serious interpretation of already collected information; second, optimize further monitoring; third, predict radioecological situations and control them in terms of decision-making persons.

#### A. "Radioecological meter" for individual sea dumping sites.

The essence of the project being proposed is shown on Fig. 5. All already existing information about dynamics of radionuclei concentration in water, bottom soil, benthos at different depths and different distances from the dumping sites will be folded (without losses) and reflected onto scales  $\Psi_1/\Psi_2$  or  $T$ . That will allow to make interpretation of radioecological state of the studied objects more comprehensive due to system effect of the whole complex of radioecological information. Observing evolution of reference points SP, S, or  $St_T$  one could reveal and predict any possible pathologies.

Studying sensitivity of drift of reference points along the scale against variation of measured parameters by numerical experiments (both deterministic and stochastic) it would be possible to optimize strategy of further monitoring thus abandoning measuring of non-informative and hardly identified parameters and estimate effectiveness of any proposals on radioecological state control. Results can be presented in form of electronic maps containing generalized indexes or, optionally, any of parameters which can be easily restored from the folding.

#### B. Radioecological meter for ranging radioecological states of dumping sites for large regions and world ocean in whole.

Fig. 6 illustrates the essence of the proposal. All the existing information on individual dumping sites can be reflected onto single scale which could be exploited as described above.

It must be emphasized that such EM would make possible to compare environmental and radioecological risks having different nature because not only radionuclei concentrations would be taken into account, but also peculiarity of their distribution and migration in ecological chains.

All mathematical and informational aspects of EM creation (analysis of data structure for sea dumping sites, monitoring optimization, ecological and environmental interpretation, visualization etc.) could be done by the authors in Russian Research Center "Kurchatov Institute", Moscow, in frames of general programme formed by Woods Hole Oceanography Institute.

EM creation for an individual disposal site (local EM) requires about four man-years; for global one — about ten man-years provided that all databases on radioecological information already exist or would be created by other explorer groups.

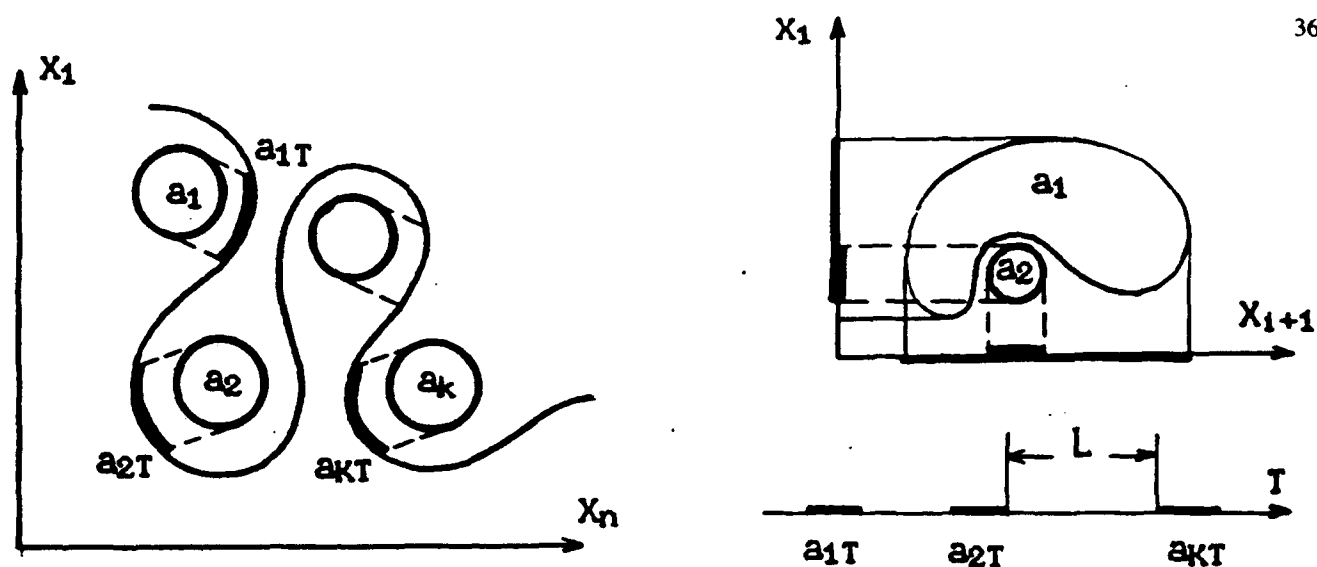
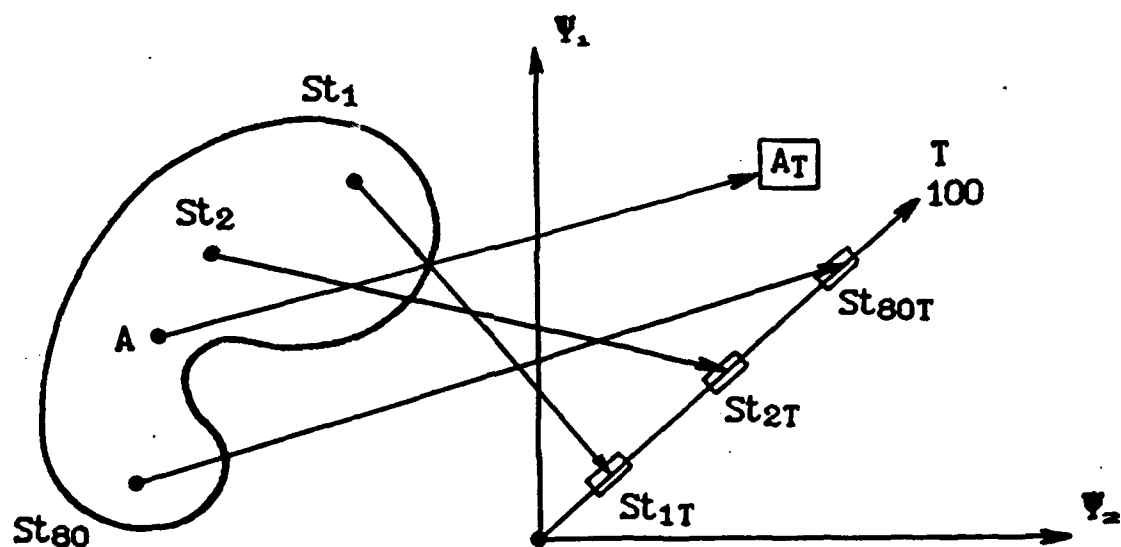
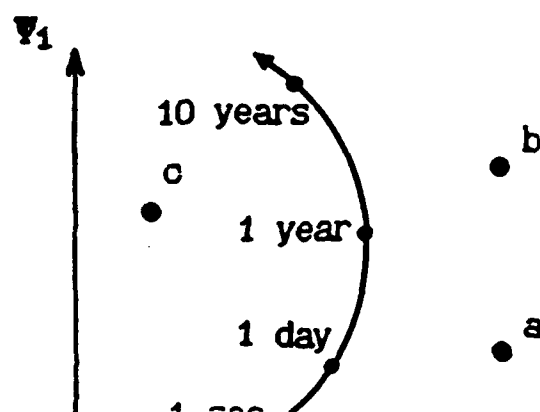


Fig. 1



$$St_1 \left\{ 110M_{Ag}; 134, 137Cs; 89, 90Sr; 106Ce; 14C \right\}$$

Fig2



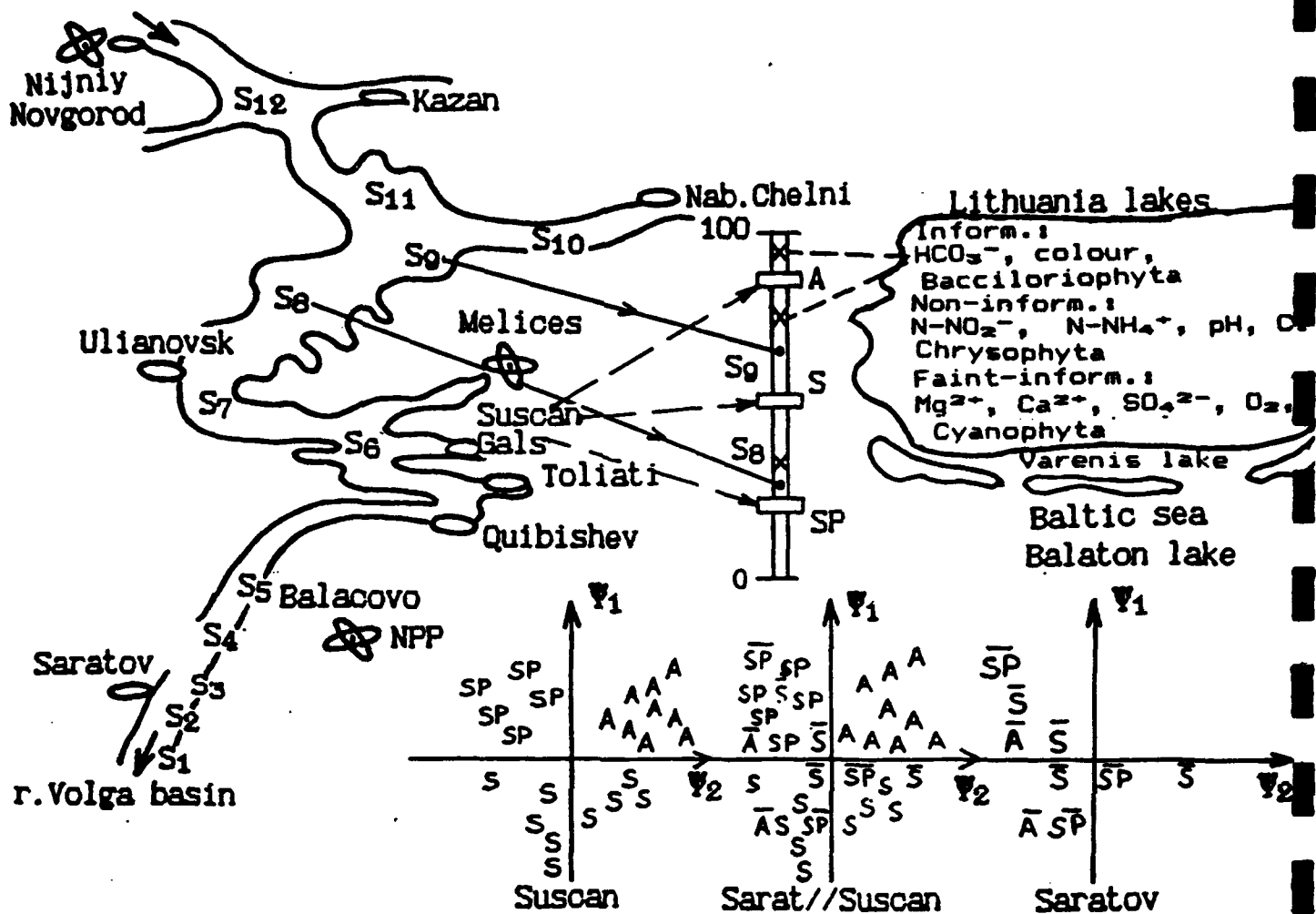


Fig. 4

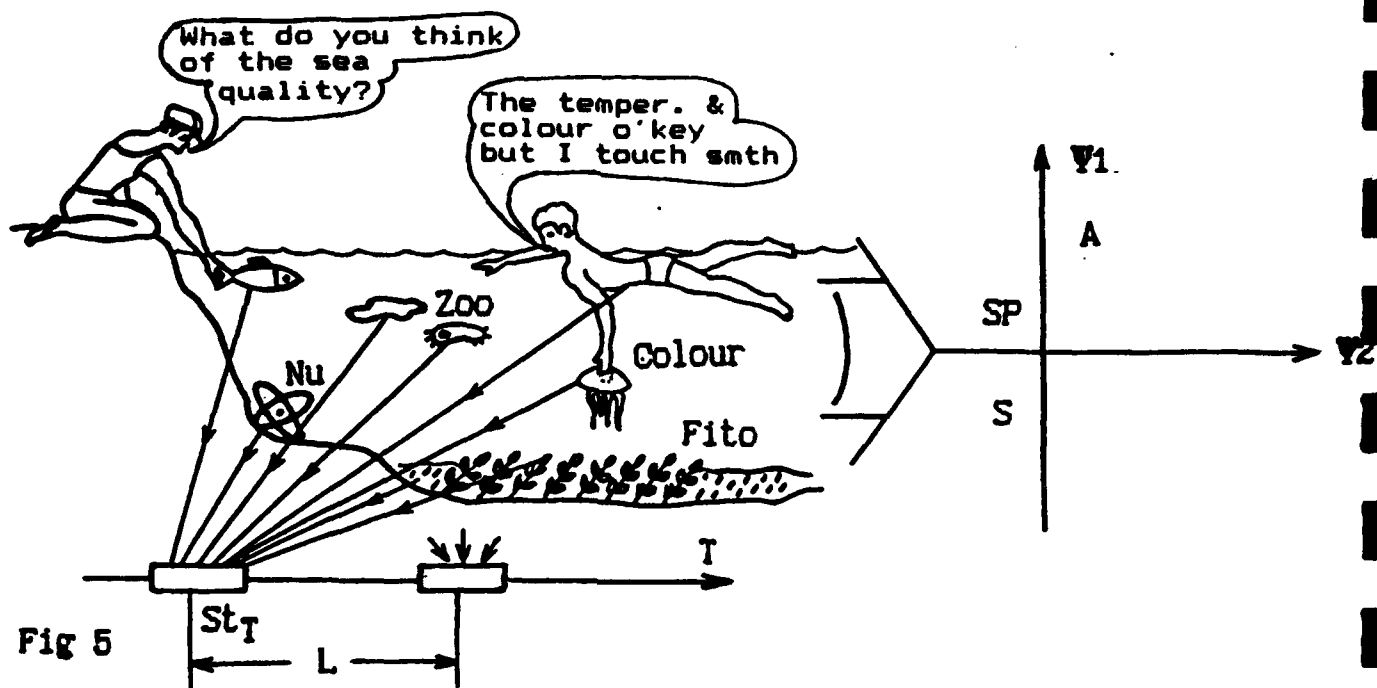


Fig 5

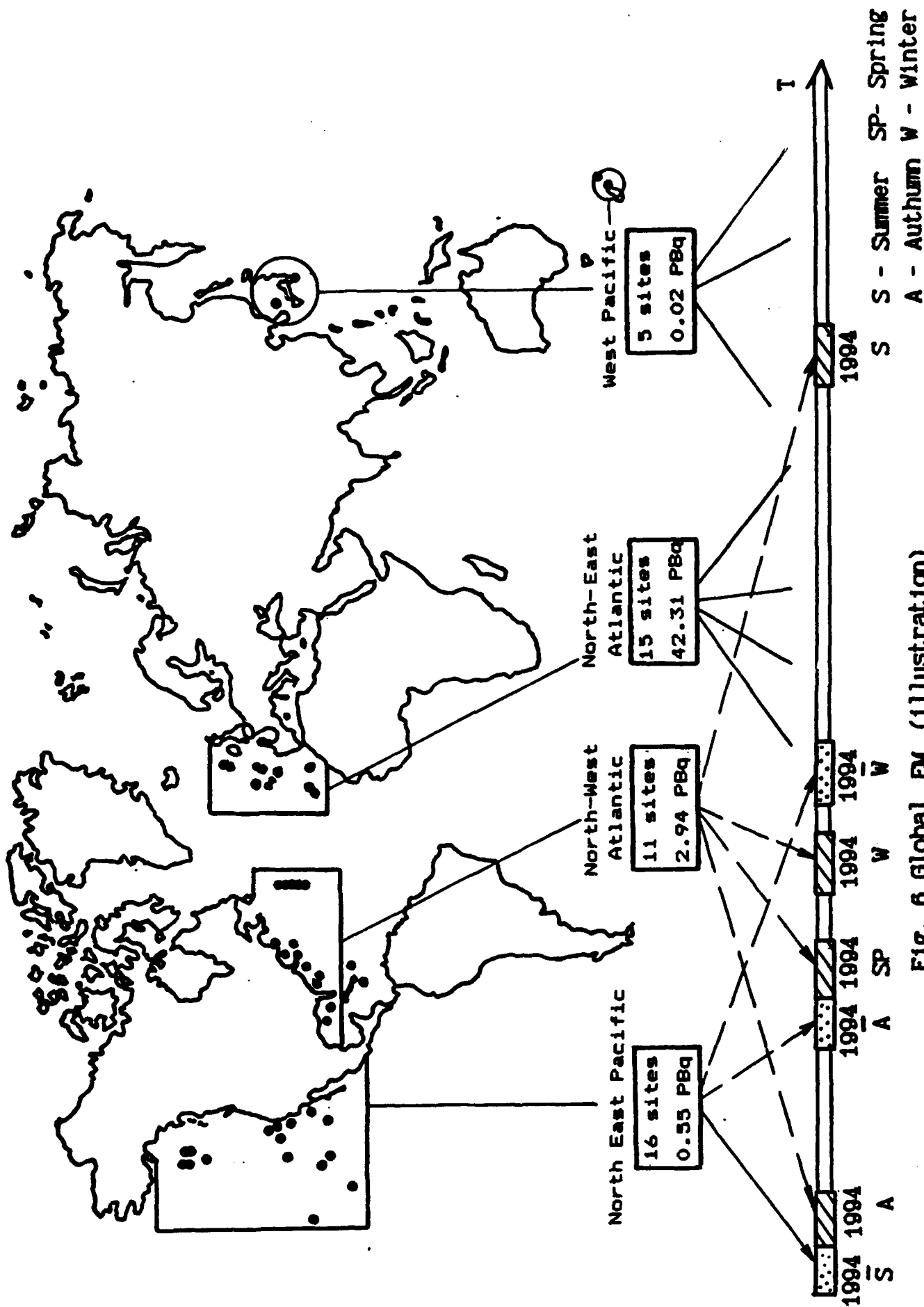


Fig. 6 Global EM (illustration)





## **The application of offshore drilling technology to the entombment of artificial sources of radioactivity on the sea floor**

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### **Abstract**

If research into the effects of artificial sources of radioactivity on the sea floor indicates that intervention is necessary, offshore drilling technology is available to provide some of the solutions. Using the Ocean Drilling Program's drillship SEDCO/BP 471 as a model, the paper describes the ability of a drillship to build structures of steel, mud and cement on the sea floor. Such structures could be used to bury artificial sources of radioactivity and slow down the release of radionuclides into the ocean.

### **Introduction**

For more than ten years in the nineteen seventies and eighties, scientists from many countries studied the feasibility of disposing of high-level radioactive waste in the sediments of the ocean floor. The research concentrated on a small number of study areas in the Atlantic and Pacific Oceans. Detailed guidelines for defining suitable disposal sites were written. For political reasons this research program was abandoned after more than \$200 million had been spent. Yet many of the scientists involved believe that, if high-level radioactive waste is to be disposed of, the sediments of the ocean floor might be the best place on earth in which to dispose it.

Since this effort to search for optimum conditions for disposing of radioactivity in the oceans, it has become clear that many nuclear submarines and reactors of the former Soviet Union have been dumped onto the sea floor in far from ideal conditions. In addition, nuclear submarines of both the USA and the USSR have accidentally sunk.

Essentially three options are available for dealing with the problem of an artificial source of radioactivity lying on the sea floor:

1. Raising it. This might be a very expensive solution which could result in an increased release of radionuclides into the ocean and be hazardous to the personnel involved. The problem of finding a new disposal site must also be addressed.
2. Letting it be.
3. Entombing it in a structure of steel, mud and cement to slow down the release of radionuclides into the ocean.

This paper concentrates on the application of offshore drilling technology to achieving the third option.

#### Drillship SEDCO/BP 471

In 1984 the drillship SEDCO/BP 471 underwent major modifications to convert her to a scientific role and in January 1985 began operations for the internationally-funded Ocean Drilling Program (ODP). The vessel is on long-term charter from Sedco-Forex, one of the world's largest drilling companies, to Texas A&M University, the Science Operator of ODP. To the scientific community which uses her she is generally known as the *JOIDES Resolution* (Foss, 1985).

SEDCO/BP 471 was built in Halifax, Nova Scotia and completed in 1978. She was designed for deep water operations for the offshore oil industry and in her original role was equipped for dynamically positioned riser drilling in water depths of up to 6000 ft (1829 m). For the Ocean Drilling Program she no longer uses a riser, but carries a much longer drill string. In riserless mode she can operate in water depths of up to 27,000 ft (8230 m) and suspend a drill string of up to 30,000 ft (9144 m) in length. To date operations have been carried out in water depths ranging from 38 m (ODP Leg 143, Site 870) to 5969 m (ODP Leg 129, Site 802).

This paper draws on the experience of ODP and takes the SEDCO/BP 471 as a model to demonstrate the capability of a drillship for building structures on the sea floor. Such a capability could be used to bury artificial sources of radioactivity and slow down the release of radionuclides into the ocean. Worldwide there are about a dozen drillships of broadly similar capability to the SEDCO/BP 471 which could carry out such work. Specifications of the SEDCO/BP 471 and of the drill rig she carries are given in Table 1.

Length	470 ft	143 m
Beam	70 ft	21.3 m
Draft	24.5 ft	7.5 m
Displacement, maximum	18600 sh. tons	16874 tonnes
Cruising speed	12 knot	
Shaft power, cruising	9000 hp	6.7 MW
Thruster power, (12 thrusters)	9000 hp	6.7 MW
Electrical generating capacity	13.5 MW	13.5 MW
Water distilling capacity	650 bbl/day	103 tonnes/day
Fuel storage	4000 sh. tons	3629 tonnes
Drill water storage	1500 sh. tons	1361 tonnes
Liquid mud storage	4100 bbl	652 m <sup>3</sup>
Bulk mud/cement storage	13600 ft <sup>3</sup>	385 m <sup>3</sup>
Mud pumps (two pumps)	1700 hp each	1.27 MW each
Rig Hoisting capacity	600 sh. tons	544 tonnes
Maximum water depth, drilling ops.	27000 ft	8230 m
Maximum drill string length	30000 ft	9144 m

1 Short Ton = 2000 lb = 0.9072 tonnes (metric)

**Table 1: SEDCO/BP 471 – Vessel and drill rig specifications**

### Capability of the SEDCO/BP 471

Most of the holes drilled in ODP are single-bit holes which do not involve any kind of structure on the sea floor. The depth of such holes is limited by the life of the drill bit. To achieve deeper penetrations, re-entry holes are required. About half a dozen re-entry holes are established each year, each costing roughly \$100K more in hardware (re-entry cone, several hundred meters of casing, cement) than a single bit hole. The process of re-entering a hole is shown diagrammatically in Figure 1.

A re-entry hole is a structure of steel and cement built on and under the sea floor. Several pipe trips are required to establish the structure: First the re-entry cone is lowered to the bottom and the short length of conductor casing beneath it jetted in. Typically a 14 3/4 inch (37.5 cm) hole is then drilled several hundred meters below the sea floor into firm rock and left full of mud to keep it stable. Next, several hundred meters of 11 3/4 inch (30 cm) casing are run into the hole and cemented to the wall rock. Finally the cement plug remaining in the bottom of the casing is drilled out. Coring for scientific purposes can then proceed. The process of establishing a re-entry hole takes several days of ship time in oceanic depths.

In order to perform such a task in a wide range of sea states and weather conditions, the drillship has four key features:

1. Precise dynamic positioning.
2. Ability to deploy heavy loads through a centre-well or moonpool.
3. Ability to view the sea floor beneath the drill string with TV.
4. Ability to mix and pump large volumes of cement or mud down the pipe into bottom structures.

#### Dynamic positioning

During drilling operations the SEDCO/BP 471 is maintained on station by means of its twin screws and twelve thwartship thrusters. A single acoustic beacon is dropped to the sea floor and acts as a fixed reference point for the floating drillship. Pulses of sound from the acoustic beacon are detected by a hydrophone array beneath the ship. The arrival times of these pulses are processed by a computer, which determines how the ship is moving and controls the ship's screws and thrusters to maintain her in the same place. The dynamic positioning (DP) system on the SEDCO/BP 471 is able to keep the vessel stationary within one percent of water depth in almost all except extreme weather and sea conditions.

The shallower the water the greater the bending of the drill string caused by a given offset of the vessel. The demands on the precision of the DP system are therefore greatest in shallow water. On ODP Leg 143 in 1992, a test of the DP system was performed at Site 670 in 38 m of water in the lagoon of Annewetak Atoll and showed that the vessel could be maintained on station within about four percent of water depth (less than 2 m offset) in fluctuating currents of up to 0.5 kt. For this very shallow water test, a taut wire system was used to measure the offset of the drillship instead of the acoustic system used in deeper water. (Shipboard Scientific Party, 1993).

#### Deployment of heavy loads

The derrick, drawworks and moonpool of the SEDCO/BP 471 allow the ship to deploy large and heavy loads to the sea floor in most weather conditions. Mounted on the travelling block of the derrick, the Western Gear heave compensator is rated at 400 short tons (363 tonnes). Use of the heave compensator reduces the amplitude of the heave at the bottom of the drill string to some 25 percent of its surface value and allows large objects to be placed relatively gently into boreholes or onto the sea floor.

The heaviest object deployed from the SEDCO/BP 471 during ODP to date was the 973 m long string of 11 3/4 inch (30 cm) casing placed in Hole 765D on Leg 123.

In water this casing string weighed 65 tonnes. The water depth was 5714 m (Shipboard Scientific Party, 1990).

The largest individual piece of equipment passed through the moonpool was the Hard Rock Guide Base deployed on Leg 106 which measured 5.2 m x 4.8 m x 3.4 m high and weighed about 20 tonnes in air (Shipboard Scientific Party, 1988).

The maximum payload which can be supported with the ODP drill string as a function of the length of the string (or the water depth) is shown in Figure 2. A dynamic overload of 20 percent due to the heave of the ship has been assumed in this calculation, but the drag effects of a large unstreamlined package on the end of the string are not included. To maximize the payload, only 5 1/2 inches (14.0 cm) pipe is used in this application down to about 2500 m. Beyond that a "tapered" drill string of 5 1/2 inches and 5 inches (12.7 cm) pipe is used. Both sizes of pipe used in ODP are manufactured from S-140 steel which has a minimum yield strength of 140,000 lb/in<sup>2</sup> (96,500 N/cm<sup>2</sup>). This is a higher grade of steel than that normally used in the oil industry.

#### TV observation of the sea floor

A TV camera lowered on an armored conducting cable down the outside of the pipe on a vibration isolated frame provides real time pictures of the sea floor to people on the drillship. This allows the ship to be precisely maneuvered in DP mode with respect to objects on the sea floor. The system is used routinely for re-entering drill holes. It is also used to conduct visual surveys of the sea floor in order to assist scientists in choosing the optimum sites for spudding holes.

The precision with which the bottom of the drill string can be maneuvered relative to objects on the sea floor was demonstrated on ODP Leg 147 in December 1992. An Ocean Bottom Seismograph (OBS) deployed on an earlier cruise by scientists from the Scripps Institution of Oceanography failed to release its sinker weight and return to the surface at the end of its recording period. The instrument, measuring approximately 1 m x 1 m x 2 m, was observed on the TV during operations at Site 894 in 3030 m water depth. Guided by the TV image, the ship was maneuvered with dynamic positioning so that a grappling hook attached to the end of the drill string engaged a 30 cm diameter "D" ring on top of the OBS. The OBS was then recovered with the drill string.

#### Use of mud and cement

A highly developed industry exists to cater for the needs of oil and gas exploration and production for drilling muds and cement. No special mud or cement has been developed for ODP; the program has relied on "off the shelf" products already in existence. Furthermore, these materials are available in bulk quantities at reasonable prices, with the exception of some of the more exotic additives.

The physical and chemical properties of drilling muds can be adjusted over a very wide range. By the use of different materials and additives, the density, viscosity, chemistry, pH and oxygen content of muds can be controlled. Some muds also have thixotropic properties.

Various types of cement are available. By adjusting its composition, the density, setting time and strength of the cement can be varied. Casing strings are usually cemented with a slurry of cement and water. It is obviously important that the cement slurry remains liquid until it has been pumped through the drill string and into the annular space between the casing and the wellbore. Oilfield cements are therefore of high quality and their behavior can be accurately predicted. The mud/cement handling capabilities of the SEDCO/BP 471 are summarized in Table 1.

### Downhole logging tools

Boreholes drilled for the oil and gas industry and for ODP are routinely logged with a range of downhole instrument packages. Among the properties measured is the gamma ray spectrum due to the presence of uranium, thorium and potassium in the rock. Neutron activation analysis is also carried out to measure the concentrations of several other elements. The Natural Gamma Tool or the Gamma Ray Spectrometry Tool without its neutron source could be used, with minor modification, to measure excess levels of radioactivity in the water column. Both of these downhole logging tools are engineered to pass down drill pipe with an internal diameter of 4 1/8 inches (10.5 cm).

### Burial of a radioactive source on the sea floor

If research into the effects of an artificial source of radioactivity on the sea floor indicates that intervention is necessary, the technology described above could be used to bury the source in a structure of steel, mud and cement in order to slow down the release of radionuclides into the water column. The method employed would be similar to that shown in Figure 3.

First a coffer-dam would be built around the wreck on the sea floor. This would be constructed from a number of "building blocks," each being the maximum convenient size for passing through the moonpool and requiring a single pipe trip for placement on the sea floor. Each block would be bolted or welded together above the moonpool from prefabricated pieces of steel in order to ease storage and handling problems on board the ship. The volume of each block would be of the order of 50 m<sup>3</sup>. The internal volume of the blocks could either be open to the sea, or be filled with heavy drilling mud if greater weight were required. When the coffer-dam has been completed and entirely surrounds the wreck, the seawater within it would be displaced by mud or cement pumped down the drill string. Cement would probably be the best material to use for this purpose, because once it had set it would remain in place for a very long time.

### Information required

Prior to a burial operation such as that described above, information would be required to define the nature and size of the problem. A high-resolution bathymetric map of the vicinity of the wreck, information on the nature of the rock or sediment on which the wreck lies, information about the current regime in the water column above it – would all be essential.

### Acknowledgements

I thank Glen Foss for reviewing the first draft of this paper, Matt Stahl for providing calculations on the payload capability of the ODP drill string and Michelle Curtis for help with the illustrations. The Ocean Drilling Program is funded through Joint Oceanographic Institutions, Inc. by the United States National Science Foundation. Approximately 55% of the funds supporting the program come from the USA, the remaining 45% in varying proportions from 18 other member countries.

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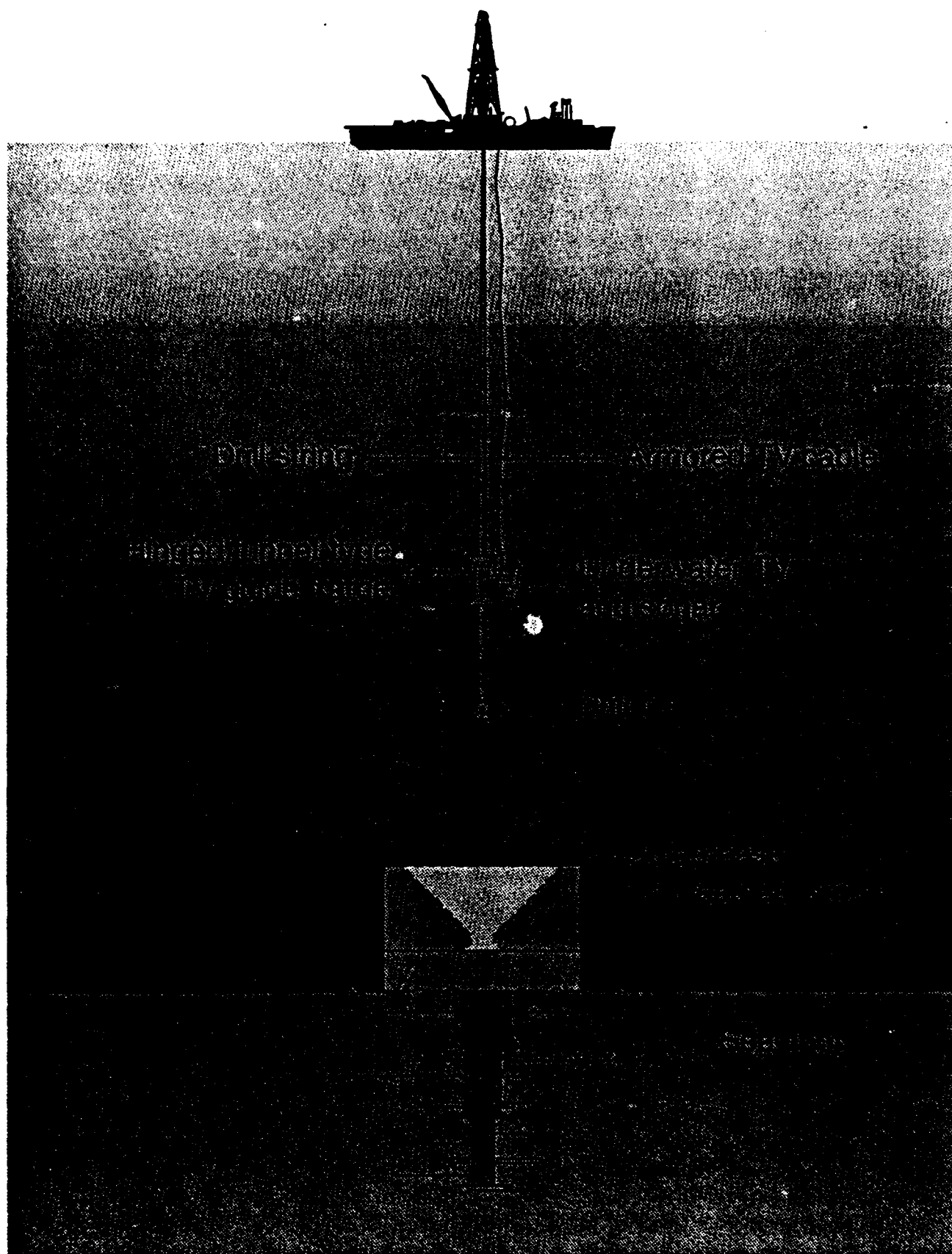


Figure 1. Worm's eye view of ocean drilling. Re-entry holes are located with TV or sonar for re-entry. Below the re-entry cone, steel casing has been run into the hole and cemented to the wall rock to stabilize the hole.



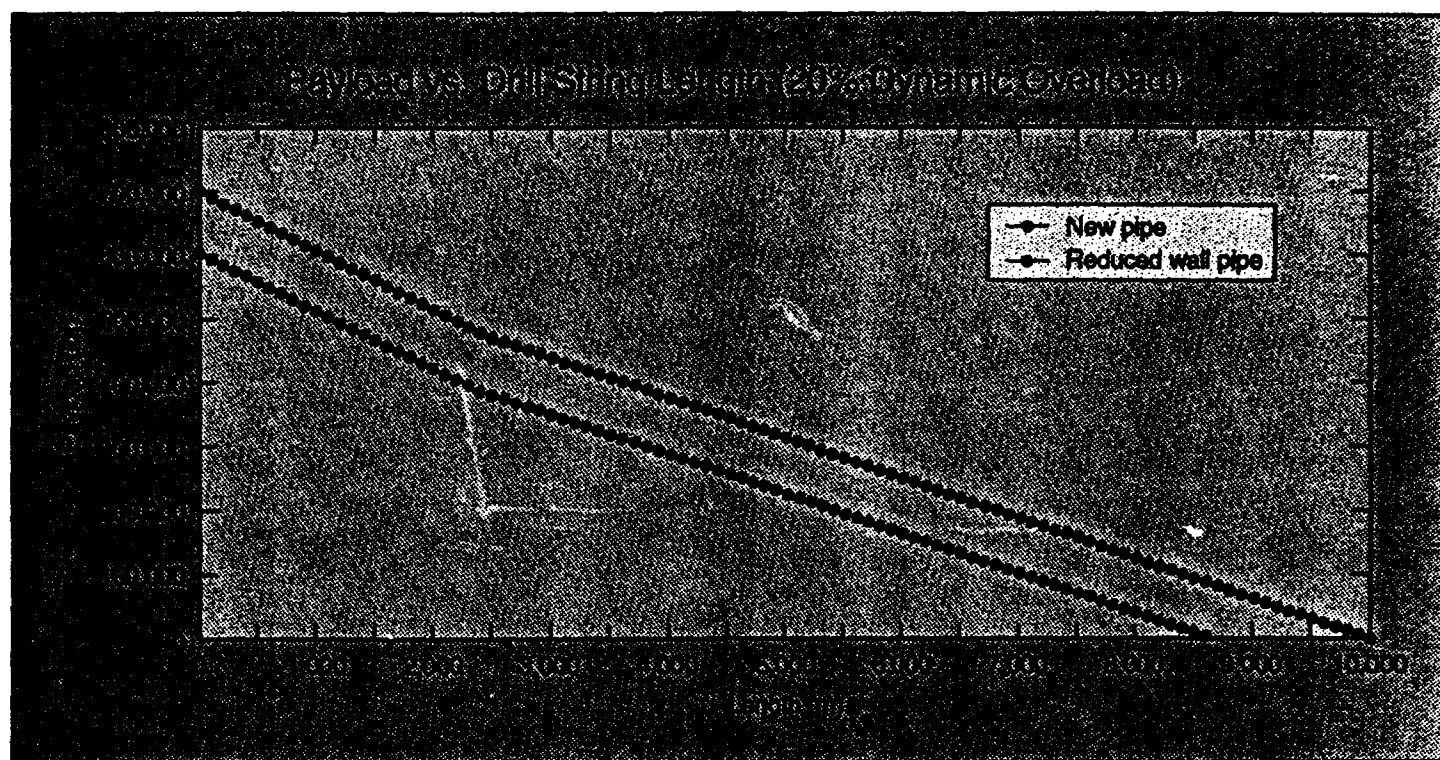


Figure 2. Graph of the maximum payload which can be supported by the ODP drill string as a function of water depth, assuming 20 percent overload due to heave. Only 5 1/2 inch pipe is used down to about 2500 m water depth. For greater depths a "tapered" drill string of 5 1/2 inch and 5 inch pipe is necessary. (2205 lb = 1 metric tonne)

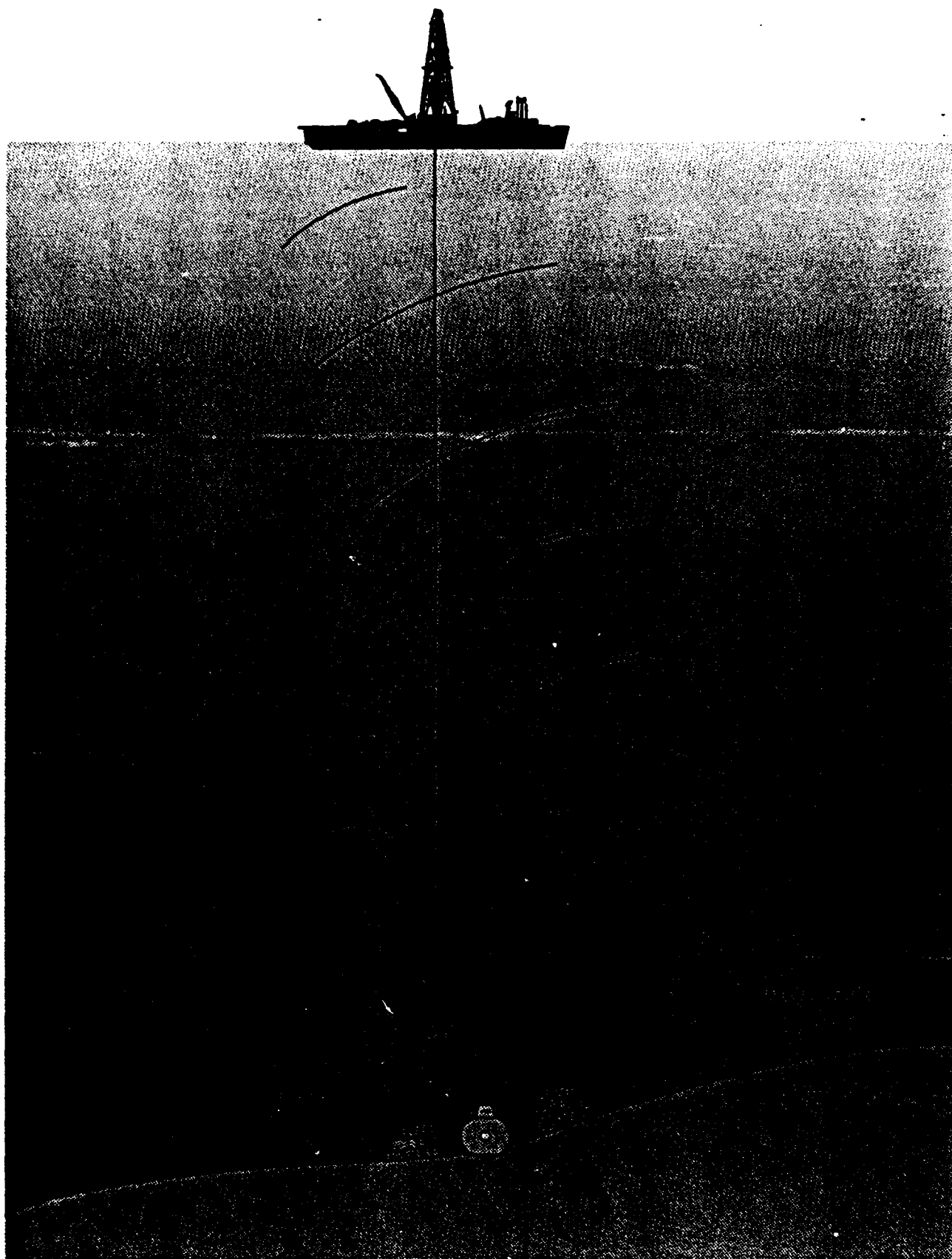


Figure 3a. Construction of a coffer-dam around a wreck on the sea floor.

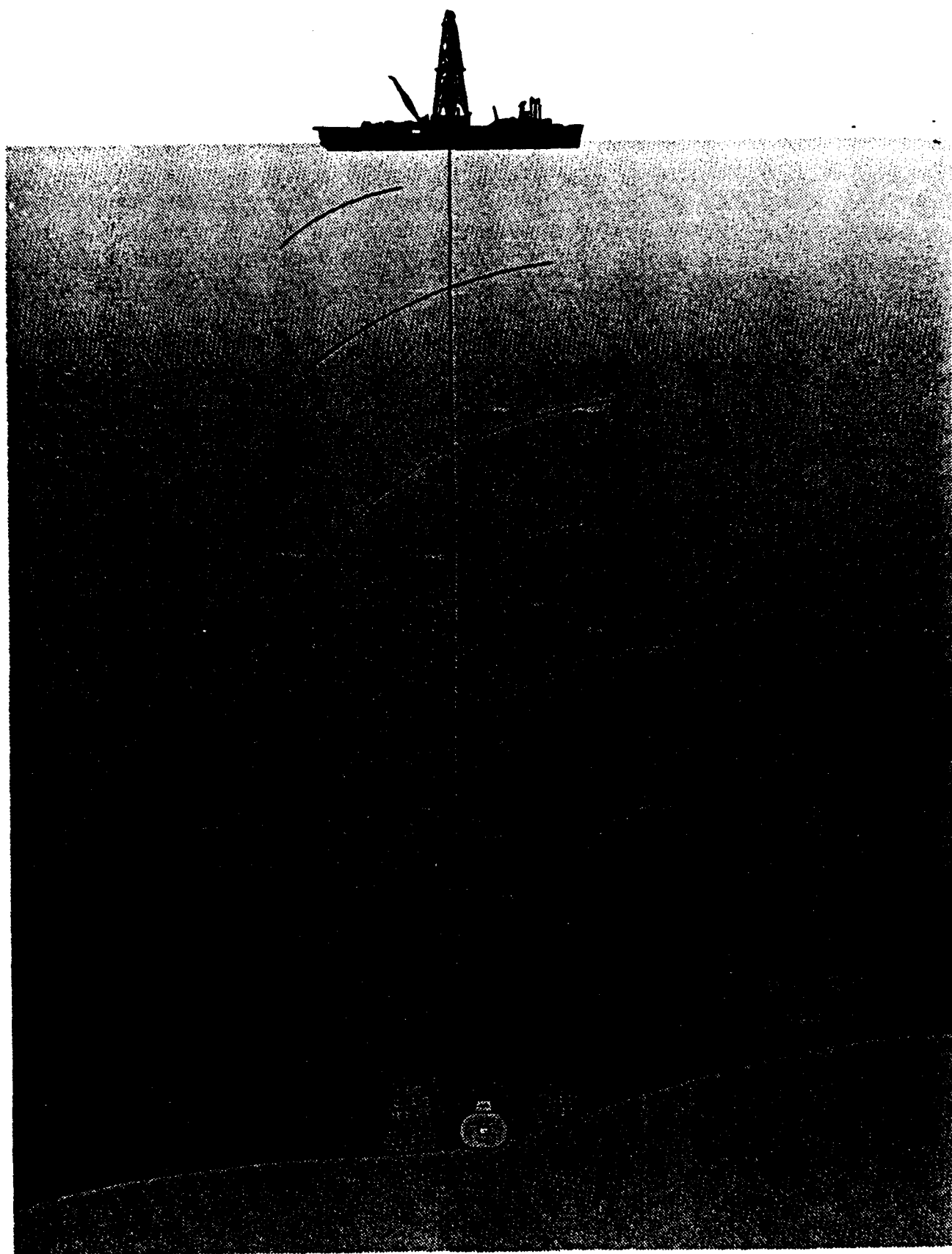


Figure 3b. Displacing water within the coffer-dam with cement or mud.



"The Radioactive Contamination of the Northern Seas:  
Approaches to the Assessment of the Impact on the  
Marine Environment and Man".

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The sources of the Northern Seas' radioactive contamination with technogenous radionuclides may be the following:

- radioactive fallouts from atmospheric nuclear weapon tests,
- radioactive fallouts from the Chernobyl NPP accident,
- spent nuclear fuel reprocessing liquid radioactive waste dumping,
- sea bottom disposal of solid radioactive wastes,
- wastes arising in operation of nuclear ships and submarines,
- radionuclide transport with continental river discharge,
- surface tests of nuclear weapons.

Some of these sources, which could be considered as local (e.g. spent nuclear fuel reprocessing liquid low-level wastes dumping in the Irish Sea(1), solid radioactive waste dumping in the Barents and in the Kara Sea), has a much broader character by spreading in the sea. Thus, for example, the conducted by us investigations of the spatial spread of Cs-134 and Cs-137, having the evident waste origin, in the North and Baltic seas shows(2)(Figs. 1, 2), that the radiocesium contamination spread during 4 years not only on considerably aquatory of the North Sea, but also penetrated into such semi-closed water reservoir as the Baltic Sea. According to data by I. I. Nikitin et al(3), up to 40% of Cs-137 in the water body of the 0-250 m layer of the Arctic Ocean have industrial origin.

Location investigations of the Sellafield waste dumping transport show that, besides Sr-90 and cesium isotopes, such radioecologically hazardous radionuclides as plutonium and transplutonium elements are carried away from the Irish Sea into the open ocean.

A potential hazard of the sea medium plutonium contamination may be from sunken nuclear-powered submarines(SSN). Thus, the SSN "Komsomolets", sunk on April 7, 1989 in the Norwegian Sea at a depth 1700 m, has on board a water-water type nuclear reactor and

two torpedo-rockets with nuclear ammunitions. The conducted by us investigations on the content of plutonium in adjoining the submarine sea water masses, bottom sediments and biota (Table 1, 2) do not reveal deviations from global levels characteristic for that region. In the opinion of the authors of reference (4), however, such a favourable radiation-ecological situation, probably, will not last in future due to equipment corrosion in sea water, which requires an organization of careful deep-water radiation monitoring.

Places of solid radioactive waste disposal (e.g. disposal of reactors with unloaded cores on the bottom) can be also considered as potentially hazardous from the viewpoint of the sea medium plutonium contamination.

Although the spectrum of radionuclides being part of radioactive wastes is quite broad (fission products, induced radionuclides, transuranium elements) and many of these radionuclides are a serious threat for sea ecosystems, however, the greatest public concern is caused by penetration of plutonium into sea medium. This is connected with the hazardousness of plutonium on the long-term scale, because the plutonium half-life is so great, that all engineer barriers protecting radioactive waste from impact of sea water will be fully destroyed by corrosion during this period, and plutonium will penetrate into the environment.

These barriers can remain intact during the life-time of such radionuclides as Cs-137, Sr-90, Co-60 etc., which will prevent their transition into the sea medium.

Evidently, a real assessment of the danger for the sea environment and, in the end, for man from plutonium contamination of seas and oceans requires a full enough information about the distribution, the forms of existence and migration, the mechanisms of accumulation on the sea bottom and of fixation in bottom sediments, of this radionuclide as one of the most long-lived and toxic components of radioactive waste.

Let us consider the main conclusions of investigations published in literature by now and devoted to the behaviour of plutonium in the sea medium.

The published by now comparatively few experimental data testify that the main quantity of plutonium in water systems is accumulated in bottom sediments (5, 6-12), as it is efficiently and quickly absorbed by suspended matter and deposits onto the bottom.

The accumulation coefficients (AC) of plutonium vary within broad

limits, decreasing in the sequence: bottom sediments > plankton, algae > corals, fishes (5). The distribution coefficients (DC) of plutonium are much influenced by its oxidation state. It is shown by example of deep-water sediments, that the DC of Pu (3+4) is by 2-3 orders higher than that of Pu (5+6) (19), i.e. it can be more mobile in the hydrosphere in higher oxidation states. Plutonium's association with suspended matter and bottom sediment is characteristic not only for sea, but for river and lake reservoirs as well (bottom sediments of these reservoirs are similar to bottom sediments of the Arctic seas). Thus, we found in studying the plutonium distribution in the Pripyat and the Dnieper rivers (Table 3) after the accident at the Chernobyl NPP, that plutonium is efficiently accumulated by suspended matter (Table 4) and bottom sediments (Table 5) of these rivers (20), its behaviour being much similar to the behaviour of such an easily hydrolysed element as cerium.

The plutonium that has come to the bottom sediments is tightly fixed in the sediment structure, not revealing a pronounced tendency (under oxidation conditions characteristic for river systems and well ventilated sea reservoirs) to back transition into surrounding water body. In studying the plutonium leachability from the bottom sediment of different rivers, we showed that under the action of river water and distilled water not more than 3% of Pu went into the aqueous phase (Table 6).

It is assumed that in ocean and sea waters plutonium exists in several physico-chemical forms: in the sorbed on the suspended matter, the colloidal and, to a very a very small degree, in the soluble form. A considerable part of plutonium in suspensions is associated with organic components of suspended matter (5).

Plutonium in the sea medium is efficiently absorbed by hydrobionts ( $AC \cdot 10^5 - 10^6$ ). It is presumed that plutonium absorption may go by the metabolic way, by surface adsorption and swallowing contaminated food, suspended matter and bottom sediments by benthos organisms (21,22). A dependence is noticed of plutonium absorption on its existence form and oxidation state.

A decrease of the AC by hydrobionts with an increase of the trophic level is characteristic for plutonium (5,23). The prevailing mechanism of plutonium bioaccumulation by lower level hydrobionts (phytoplankton, zooplankton) is surface adsorption (22).

It is assumed that due to an extremely low concentration of plutonium in aquatic systems, insufficient for forming its own

compounds(24), its behaviour in the sea medium will be much determined by nonisotopic macrocarries(iron, manganese)(5).

Summarizing the aforesaid, one may come to a conclusion, that published in literature data concerning plutonium in the sea medium are fragmentary, insufficient for a completar and deeper study of its behaviour in the sea medium, accumulation in bottom sediments, bioaccumulation, remobilization in sediments, etc. One of the reason of it is that the levels of plutonium content in the sea medium's components are very low, which often makes it imposible not only to determine its content with a required precision, but to a still greater extent does not allow to study its existence forms and migration processes.

At the present level of our knowledge the above-mentioned circumstances interfere with sure forecasts of plutonium' behaviour under these or those conditions of the sea medium. For this reason it seems to us quite desirable to obtain any information supplementing the direct study of plutonium behaviour in the sea medium, even if this information is indirect.

One of the sources of such information may be obtained by now numerous data on the distribution, the physico-chemical behaviour, the existence forms and the mechanisms of migration in the sea medium, of such long-lived radioinuclides as Th-230, Th-232 and Pa-231(25). In many respects these elements are chemical analogues of plutonium, and their use as a kind of tracers of plutonium's behaviour in the sea medium is based on an assumption (26) that these elements behave much similary in the specifical physico-chemical situatuion of sea and oceans. Besides, the concentrations of natural long-lived chemical analogues of plutonium in the sea medium components is tens of times higher than the global level of plutonium concentration, which essentially increases the expressness of radiochemical investigations in seas and oceans.

A premise for the assumption about the closeness of the behaviour of plutonium in aquatic systems on the one hand, and of protactinium and isotopes of thorium on the other hand, were comparative investigations of the physico-chemical state of microquantities of plutonium, thorium and protactinium in aqueous solutions, carried out at the Radium Institute in 1950s- 1960s(27). The main conclusions of those investigations are briefly summarized in Table 7. As follows from the data cited in Table 7, the behaviour of all the three elements in aqueous solutions is practically the



same in quite a broad pH region. An increase of the solutions' ionic strength (the ionic strength of sea water is known to be quite high) causes an intensification of the mutual coagulation of colloids and efficient extraction of all three elements from the solutions into the solid phase.

Specific properties of the sea medium (the constancy of the salt composition, the ratio between its main components, the narrow limits of pH variation, etc.) verify the validity of the hypothesis about the closeness of three considered elements' physico-chemical behaviour in seas and oceans.

Since late 1950s for many years we have been conducting radiochemical investigations in seas and oceans, aimed at studying the behaviour of radioelements therein (25). At the very beginning special attention was paid to studying the behaviour of such natural radionuclides as U-234, 238; Th-230, 232; Pa-231, Ra-226, the ratios between which in columns of sea and ocean sediments are used for determination of the sediments' absolute ages and their sedimentation rates. The distribution of the said radionuclides in waters, suspensions, surface sediments and columns of sea and ocean sediments, was studied on the basis of materials sampled by complex Soviet expeditions in Antarctica, the Indian, the Atlantic and the Pacific oceans, as well as samples taken in inner and outer seas washing the USSR. As a rule, samples of water, suspended matter, surface sediments and sediment columns were taken in one and the same vertical section, and Pa, U, Ra, Th-isotopes were determined in the same sample, which increased the reliability of conclusions about the mechanisms of the studied radionuclides' deposition from water to sediment. In Fig. are shown the sampling sites of the sea medium components studied by us.

As seen, a considerable part of samples belongs to the southern part of the Indian Ocean where there is practically no river discharge, which essentially simplifies the study of the mechanisms of radionuclides' transition from water to sediment.

Besides the direct study of the radionuclides distribution in the sea medium components we made series of simulated experiments, using real sea waters, suspensions and sediments for studying the sorption behaviour of thorium and plutonium in those systems.

The comparative mobility of Th-230 and Pu-239, 240 was also investigated in different types of bottom sediments.

Special attention was paid to studying the accumulation of tho-

rium (Th-234) and plutonium-238 by hydrophytes (Table )( ).

The obtained by us broad enough experimental material is but a small part of the giant arsenal of experimental data on the distribution of Pa, Th-isotopes, U and Ra in the sea medium, published in literature by now. Nevertheless, on the basis of our data we formulated main conclusions drawn in the course of the above-mentioned investigations. These conclusions are briefly as follows (Table ). For comparison the main conclusions on behaviour of plutonium in the sea medium, based on a literature data survey, are summarized in the same table.

Thus, the whole complex of investigations carried out by us may be considered as a weighty argument in favour of using thorium-230, protactinium-231 and thorium-232 as a kind of tracers for a more detailed study of the behaviour of plutonium in the sea medium. Naturally, the aforesaid does not exclude the necessity of direct investigations of plutonium's behaviour in seas and oceans, equally as any other approaches promoting the progress of our knowledge in this direction.

The advantages of the suggested approach are:

1. As plutonium is a "new" element for the sea medium and has no stable isotopes, the use of elements - plutonium's chemical analogues, that have been present in seas and oceans always, is doubtlessly an important source of obtaining additional information about its behaviour in the sea medium.

2. The accession of plutonium into the sea medium proceeded during a period of time incomparable by its duration with geological, biological, etc., processes in seas and oceans, through which have passed the "always" present in the sea medium natural long-lived radionuclides. The study of these nuclides' behaviour in the sea medium at present and in the past, that has revealed, in particular, the closeness of the mechanisms of their penetration into bottom sediments with time, is a weighty argument in favour of an assumption that in the future these mechanisms will be the same. The extension of this conclusion onto plutonium (given the validity of our assumption of the closeness of the behaviour of plutonium, thorium-230, protactinium-231 and thorium-232 in the sea medium opens possibilities to forecast the

behaviour of plutonium in the sea medium in perspective.

3. The extremely low concentrations of plutonium in components of the sea medium makes practically impossible to study directly the forms of its existence in those objects. The considerably greater contents of natural radionuclides in the same objects opens perspectives for the application of classical methods of studying their states in aqueous solutions ( ).

4. The low concentrations of plutonium in the sea medium hinders a timely obtaining of information about its distribution in this or that region of the World Ocean. The higher concentrations of protactinium and isotopes of thorium in sea medium components, and the accumulated methodological experience of determining them, allow to study these radionuclides' distribution in the sea medium timely enough.

5. There are considerably more accumulated experimental data concerning the distribution of the concentrations of natural radionuclides in the sea medium, their existence forms, mechanisms of migration and penetration into bottom sediments, as well as their behaviour with time in the very structure of sediments, than those for plutonium.

6. Thorium-230, protactinium-231 and thorium-232, chosen for forecasting the behaviour of plutonium in the sea medium, must give a full enough coverage of the sources of plutonium's penetration into the sea medium and its behaviour in it. Indeed, according to our conception, thorium-230 can characterize the sea medium behaviour of hydrolysed plutonium forms where plutonium is in the 4-valent state. The same forms may be represented in the sea medium by protactinium-231, covering the 5-valent state of plutonium. To describe the sea medium behaviour of plutonium coming to seas and oceans as part of terrigenous material ( global radioactive fallout, river discharge, products of surface nuclear weapon tests), useful information may be obtained from the sea medium distribution of thorium-232.

It is evident that the information about plutonium's existence forms in the ocean is necessary for an assessment of its

impact onto sea ecosystems, because the absorption of plutonium by biota and its movement by food chains to man is directly connected with the form of its existence in sea water.

The presented above approach was developed mainly for deep-water regions of the World Ocean. With a certain correction, however, it may be used for forecasting the behaviour of plutonium in arctic seas. Thus, for example, we intend to use it (parallel with direct determination of plutonium) for a forecast of plutonium's behaviour in the wreck area of the submarine "Komsomolets". The determination of thorium-230 and protactinium-231 in bottom sediments of this area will allow simultaneously to calculate the bottom sediment accumulation rate and to assess the rate of plutonium's transfer onto the bottom in that area. Studying the vertical distribution of these radionuclides in sediments, side by side with determining the oxidation-reduction potentials in those sediments and the distribution of chemical components in them, the mineralogy, etc., will allow to assess the role of diagenesis processes in the course of secondary redistribution of these radionuclides (and plutonium) in the sediments studied. The whole complex of data on the distribution of plutonium, protactinium and isotopes of thorium in sea water components in the given region will enable, as we hope, not only to obtain knowledge about the sea medium behaviour of plutonium and its impact to the biota at present, but also to forecast the situation for the future.

Russian researches developed the criteria of assessment of significant (from ecological and sanitary viewpoints) levels of the sea medium radionuclide contamination (Tables 10,11) (30,31). Table 11 presents the recommended permissible concentrations in sea water of plutonium, some fission products and induced radionuclides, developed as applied for the Arctic seas, taking into account various applications of the sea medium and its bioproducts.

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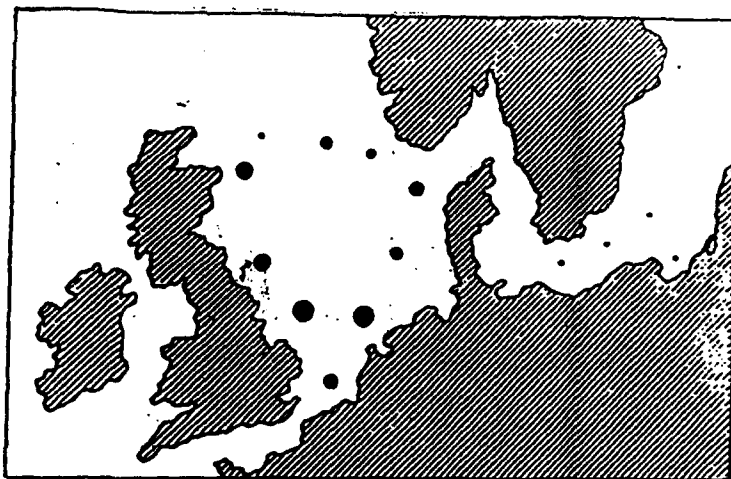


FIG. 1. DISTRIBUTION OF Cs-137 IN SURFACE WATERS OF THE NORTHERN SEA AND THE BALTIC (1975-1976 Y.). CIRCLE DIAMETER IS PROPORTIONAL TO CONCENTRATION.



FIG. 2. DISTRIBUTION OF Cs-137 IN SURFACE WATERS OF THE NORTH SEA AND THE BALTIC SEA (1979 Y.). CIRCLE DIAMETER IS PROPORTIONAL TO CONCENTRATION.

TABLE 1. PLUTONIUM CONTENT IN BOTTOM SEDIMENTS AND BENTHIC ORGANISMS OF THE NORWEGIAN SEA.

SAMPLE INDEX	Pu(239;240); Bq/Kg.	Pu-238, Bq /Kg.
<b>BOTTOM-DREDGE SAMPLES</b>		
2508	0.36±0.05	0.20±0.04
2519	0.11±0.02	0.08±0.02
2525	0.21±0.03	0.15±0.03
2542	0.12±0.01	0.08±0.01
<b>BENTHIC ORGA- NISMS</b>		
OPHIUROIDAE	0.033±0.006	<0.001
FORAMINIFERA	0.049±0.003	<0.007

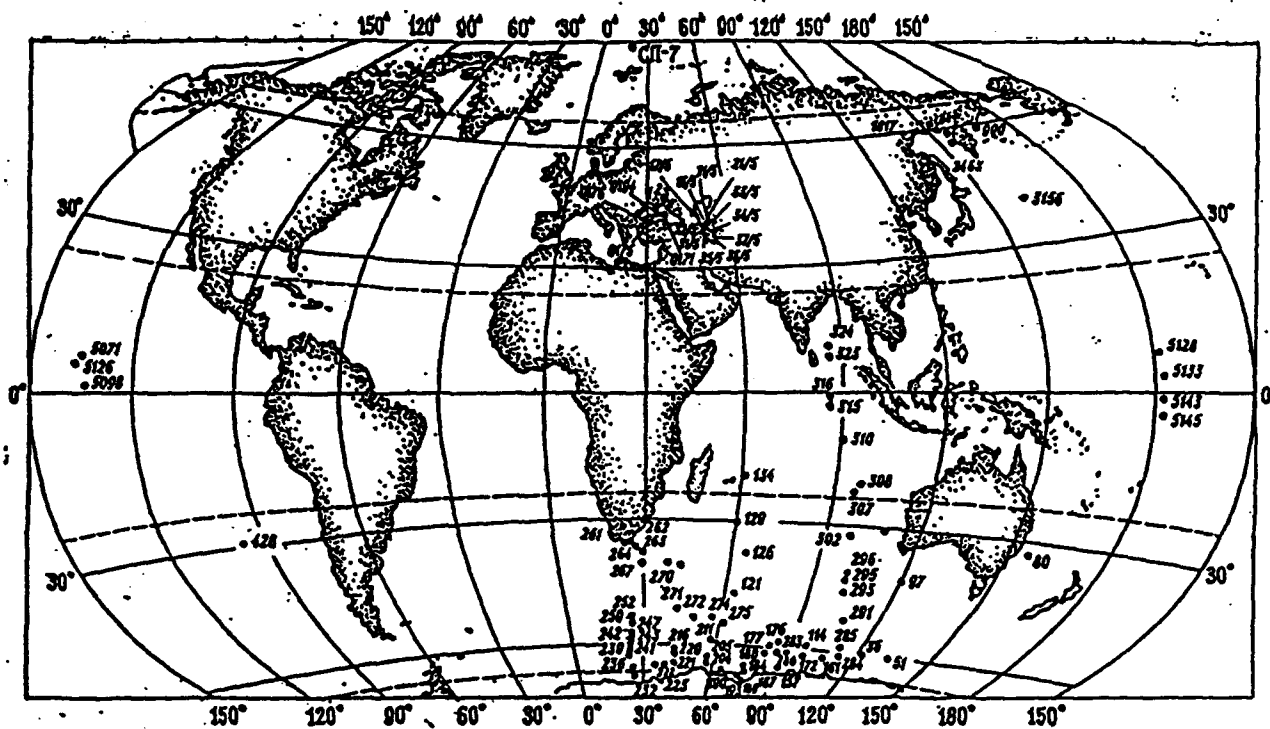


FIG. 3. LOCATION OF WATER, SUSPENDED MATTER  
AND SEDIMENT SAMPLING SITE IN THE WORLD OCEAN.



TABLE 2. PLUTONIUM CONTENT IN WATER SAMPLES IN  
THE SSN "KONSOMOLETS" SINKING AREA.

NUMBER OF STATION	SAMPLING DATE	SAMPLING HORIZON, M	Pu-(239,240), Bq/m <sup>3</sup>	Pu-238, Bq/m <sup>3</sup>
2494	20.08.91	0	0.007	0.003
2494	20.08.91	1162	0.013	0.003
2512	25.08.91	0	0.006	0.001
2512	25.08.91	1688	0.008	-
2530	28.08.91	1672	0.012	<0.0016

Pu-(239,240) determination error: 30%.

Pu-238 determination error: 50%.

Table 3: CONTENT OF PLUTONIUM-238 AND PLUTONIUM 239,240 IN SAMPLES OF UNFILTERED  
WATER TAKEN FROM THE RIVER PRIPYAT AND THE KIEV RESERVOIR IN 1986 AND 1987

Sample index	Sample collection point	Collection date	1986			1987				
			Pu-238 Bq/m <sup>3</sup>	Pu-239,240 Bq/m <sup>3</sup>	Pu-238, Pu-239,240 Bq/m <sup>3</sup>	Collection date	Pu-238 Bq/m <sup>3</sup>	Pu-239,240 Bq/m <sup>3</sup>	Pu-238, Pu-239,240 Bq/m <sup>3</sup>	
RIVER PRIPYAT										
1.	Chernobyl town, landing stage	-	-	-	-	21.05	0.70 ± 0.12	0.94 ± 0.14	0.74	
1 <sup>a</sup> .	Chernobyl town, landing stage	17.07	12.0 ± 3.0	46.0 ± 4.0	0.60	-	-	-	-	
2.	Above Chernobyl town	16.07	0.5 ± 0.2	0.6 ± 0.2	0.05	-	-	-	-	
3.	Chernobyl town, creek	-	-	-	-	03.06	0.17 ± 0.04	0.25 ± 0.04	0.06	
4.	Buoy 210	17.07	0.7 ± 0.2	1.2 ± 0.3	0.60	04.06	2.6 ± 0.30	4.7 ± 0.17	0.55 ± 0.05	
5.	Buoy 211	17.07	17.0 ± 2.0	46.0 ± 4.0	0.36	-	-	-	-	
6.	Buoy 219	-	-	-	-	16.05	0.14 ± 0.04	0.51 ± 0.04	0.61	
7.	Amur	16.07	3.0 ± 0.3	5.1 ± 0.5	0.36	25.05	1.9 ± 0.20	3.5 ± 0.25	0.54	
KIEV RESERVOIR										
8.	Sukholishche	19.07	-	0.3 ± 0.2	-	26.05	0.7 ± 0.15	1.2 ± 0.10	0.56	
9 <sup>a</sup> .	Sukholishche	19.07	20.0 ± 2.0	27.0 ± 3.0	0.75	-	-	-	-	
9.	Cioborsha	-	-	-	-	22.05	0.35 ± 0.15	0.4 ± 0.06	0.67	

<sup>a</sup> samples of benthic water

Table 4: RESULTS OF MEASURING THE SPECIFIC ACTIVITY OF PLUTONIUM-239,240 IN SAMPLES OF  
RIVERS SUSPENSIONS FROM THE PRIPYAT AND DNIEPER  
(May and June 1987)

Sample index	Collection point	Volume of water l	Suspension concentrat. g/m <sup>3</sup>	Specific activity Bq/kg suspension		Pu-239,240 Ce-144
				Pu-239,240	Ce-144	
1.	Buoy-203	200	21	57	1.02 E 4	3.5 E -3
2.	Buoy-210-212	300	31	155	7.97 E 4	1.9 E -3
3.	Buoy-219	200	31	31	0.90 E 4	3.4 E -3
4.	R. PRIPYAT Below pontoon bridge	200	42	20	0.57 E 4	3.4 E -3
5.	landing stage	155	47	19	1.06 E 4	1.8 E -3
6.	landing stage at Chernobyl	200	47	33	3.59 E 3	5.8 E -3
7.	Below pontoon bridge at entrance to quarry	200	41	238	4.34 E 4	3.8 E -3
8.	recess opposite mouth of R. Uzh	200	47	64	1.81 E 4	3.5 E -3
9.	buoy-40. Kiev reservoir	30	32	14	4.69 E 3	3.0 E -3

Table 5: SPECIFIC ACTIVITY OF PLUTONIUM-239,240 IN SAMPLES OF BED SEDIMENTS FROM THE RIVERS PRIPTAT AND DNIPEK (May-June 1987)

Sample collection point	Specific activity Bq/kg		Pu-239,240 Co-144
	Pu-239,240	Co-144	
Buoy at Chernobyl creek	130 ± 13	7.26 E 4	1.6 E -3
Creek opposite mouth of S. Dzh (0-3 cm)	310 ± 30	1.46 E 5	1.9 E -3
S. Dzh (3-6 cm)	370 ± 30	1.47 E 5	1.6 E -3
S. Dzh (6-9 cm)	330 ± 30	1.97 E 5	1.7 E -3
Creek opposite mouth of S. Dzh	810 ± 80	4.22 E 5	1.9 E -3
Buoy-231 (bayou)	230 ± 20	1.03 E 5	1.7 E -3
Buoy-217	100 ± 20	7.91 E 4	2.3 E -3
Buoy-203	940 ± 30	2.58 E 5	2.1 E -3
Buoy-40	23 ± 5	1.24 E 4	1.9 E -3
Buoy-15	29 ± 6	1.58 E 4	1.8 E -3
Buoy-01	15 ± 3	7.69 E 3	1.7 E -3
S. DNIPEK Kusakovsky channel	1.6 ± 0.4	0.40 E 3	0.4 E -3
Malveyevsky creek	7.4 ± 1.4	3.52 E 3	2.1 E -3
Obolonsky creek	26 ± 6	1.43 E 4	1.6 E -3

TABLE 6 . THE LEACHABILITY (%) OF PLUTONIUM(239,240) FROM BOTTOM SEDIMENTS.

SAMPLE INDEX	WATER	AMMONIUM	HCL, 6M/L
		ACETATE, 1M/L	
1	1.5	7.0	47
2	0.9	2.7	74
3	1.7	3.0	67
4	2.6	3.0	76
5	2.6	4.4	54
6	1.0	0.5	61

TABLE 7. A COMPARATIBLE STUDY OF PHYSICO-CHEMICAL STATE OF PLUTONIUM, THORIUM AND PROTACTINIUM MICROQUANTITIES IN AQUEOUS SOLUTIONS.

PLUTONIUM	THORIUM	PROTACTINIUM
IN pH REGION OVER 3 FORMS HYDROLYSIS PRODUCTS OF DIFFERENT POLYMERIZATION DEGREE: MONOMER PRODUCTS, POLYMERS WITH SMALL MOLECULAR MASS, COLLOIDAL POLYMERS WITH GREAT MOLECULAR MASS. THE MOST STABLE FORM Pu(IV), EFFICIENTLY SORBED BY IMPURITIES (e.g. NEGATIVELY CHARGED SUSPENSIONS), MAY BE CAPTURED BY THEM BY COLLOIDS MUTUAL COAGULATION MECHANISM. LESS SUSCEPTIBLE TO SORPTION HIGHER OXIDATION FORMS: Pu(V) AND Pu(VI).	SUBJECT TO STRONG HYDROLYSIS IN SOLUTIONS, WITH FORMATION OF MONONUCLEAR COMPLEXES Th OH AND MORE COMPLEX POLYMERIZED FORMS Th <sub>2</sub> (OH) <sub>5</sub> <sup>+</sup> , Th <sub>3</sub> (OH) <sub>8</sub> <sup>+</sup> etc. IN pH REGION 4-9 IS MAXIMALLY SORBED BY FINE DISPERSED SUSPENSIONS. AT pH 4 MOST PART OF Th IS IN COLLOIDAL STATE.	AT SOLUTION pH OVER 6 IS SUBJECT TO STRONG HYDROLYSIS, WITH FORMATION OF HYDROLYSIS PRODUCTS OF DIFFERENT POLYMERIZATION DEGREE, EFFICIENTLY SORBED BY BODIES WITH DEVELOPED SURFACE. MOLECULAR AND COLLOIDAL FORMS MAY ARISE, LATTER BEING EXTRACTED FROM SOLUTIONS BY COLLOIDS MUTUAL COAGULATION MECHANISM.

TABLE 8. COEFFICIENTS OF PLUTONIUM-238 AND THORIUM-234 ACCUMULATION BY HYDROPHYTES.

SPECIES OF HYDROPHYTES	ACCUMULATION COEFFICIENTS	
	PLUTONIUM-238	THORIUM-234
BENTHOS ALGAE		
NITELLOPSIS OBTUSA	2000	9000
CHARA SP.	6800	7000
HIGHER ALGAE		
CERATOPHYLLUM DEMERSUM	3300	2000
MYRIOPHYLLUM SPICATUM	2800	2000
ELODEA CANADENSIS	2900	2300

TABLE 9. CONCLUSIONS FROM EXPERIMENTAL INVESTIGATIONS OF PLUTONIUM, PROTACTINIUM AND ISOTOPES OF THORIUM IN SEA MEDIUM

PLUTONIUM	THORIUM ISOTOPES, PROTACTINIUM
1. EFFICIENTLY EXTRACTED FROM SEA WATER BY SUSPENSIONS ( $K_D = 10^3 - 10^4$ ) AND SEDIMENTS ( $K_D = 10^4 - 10^5$ ).	1. THE COMING OF Th-230 AND Pa-231 IS DETERMINED BY PROCESSES OF THEIR EXTRACTION BY FINE-DISPERSED SUSPENSIONS (FIRST OF ALL, COLLOIDAL BLOBS OF Fe AND Mn) AND BY PLANKTON ORGANISMS.
2. EVIDENT DIRECT CONNECTION BETWEEN THE CONTENT OF Fe AND Mn OXYHYDRATES IN SEDIMENTS, AND THE CONTENT OF Pu IN THEM.	2. FORMS OF Th-230 AND Th-232 EXISTENCE IN THE SEA MEDIUM ARE DIFFERENT, MOST OF Th-232 COMES TO BOTTOM SEDIMENTS AS A PART OF TERRIGENOUS MATERIALS.
3. EFFICIENTLY ABSORBED BY PHYTOPLANKTON AND BENTHIC ALGAE.	3. THE MECHANISMS OF COMING OF Th-230, Th-232 AND Pa-231 INTO BOTTOM SEDIMENTS AT PRESENT AND IN PAST EPOCHS ARE SIMILAR.
4. IN MOST CASES Pu IS TIGHTLY FIXED IN THE STRUCTURE OF BOTTOM SEDIMENTS. ITS DIFFUSION COEFFICIENT DOES NOT EXCEED $10^{-12}$ cm. sec. <sup>-1</sup>	4. Th-230, Th-232 AND Pa-231 MIGRATION IN THE BOTTOM SEDIMENTS IS NOT INDICATED EXPERIMENTALLY. THEIR LEACHING FROM SEDIMENTS IS NEGLIGIBLE.

TABLE 10. COEFFICIENTS OF MAXIMAL ACCUMULATION OF RADIOACTIVE AND STABLE ISOTOPES BY FOOD MARINE ORGANISMS.

ELEMENT	FISH	MOLLUSCS	CRUSTACEA	ECHINODERMATA	ALGAE
Ca	2	10	10	10	10
Cr	2000	500	400	-	1000
Mn	130	10000	7600	200	1200
Fe	2000	10000	2500	-	10000
Co	80	200	600	240	430
Zn	3000	10000	5000	1400	1000
Sr	2	10	10	1	60
Yt	10	18	80	-	800
Zr	150	33	100	-	1000
Nb	100	140	100	-	2000
Mo	20	100	100	-	15
Ru	100	10	10	-	2000
Cd	5	10000	1	100	1500
Sb	140	100	70	70	200
I	15	70	50	60	2000
Cs	25	60	25	10	70
Ce	25	100	100	15	700
Po	15	25	25	-	1000
Pu	25	25	45	-	2000

TABLE 11. WORKING LIMITS OF RADIONUCLIDE CONCENTRATIONS  
IN SEA WATER (Ci/L) IN LONG-TERM CONTAMINATION OF FISHING  
REGIONS.

RADIONUC- LIDES	FISHING ZONE	FISHING REGIONS			MIXED FISHING ZONE
		MOLLUSCS	CRUSTACEA	ALGAE	
Cr-51	$7 \times 10^{-10}$	$1 \times 10^{-8}$	$1 \times 10^{-8}$	$7 \times 10^{-9}$	$7 \times 10^{-10}$
Mn-54	$1 \times 10^{-9}$	$6 \times 10^{-11}$	$8 \times 10^{-11}$	$7 \times 10^{-10}$	$1 \times 10^{-10}$
Fe-55	$4 \times 10^{-10}$	$4 \times 10^{-10}$	$2 \times 10^{-9}$	$4 \times 10^{-10}$	$4 \times 10^{-10}$
Fe-59	$3 \times 10^{-11}$	$3 \times 10^{-11}$	$1 \times 10^{-10}$	$3 \times 10^{-11}$	$3 \times 10^{-11}$
Co-57	$5 \times 10^{-9}$	$1 \times 10^{-8}$	$3 \times 10^{-9}$	$4 \times 10^{-9}$	$4 \times 10^{-9}$
Co-58	$1 \times 10^{-9}$	$2 \times 10^{-9}$	$7 \times 10^{-10}$	$1 \times 10^{-9}$	$7 \times 10^{-10}$
Co-60	$4 \times 10^{-10}$	$9 \times 10^{-10}$	$3 \times 10^{-10}$	$4 \times 10^{-10}$	$3 \times 10^{-10}$
Zn-65	$3 \times 10^{-11}$	$5 \times 10^{-11}$	$1 \times 10^{-10}$	$5 \times 10^{-10}$	$3 \times 10^{-11}$
Sr-89	$1 \times 10^{-9}$	$6 \times 10^{-9}$	$6 \times 10^{-9}$	$1 \times 10^{-9}$	$1 \times 10^{-9}$
Sr-90	$4 \times 10^{-11}$	$2 \times 10^{-10}$	$2 \times 10^{-10}$	$4 \times 10^{-11}$	$4 \times 10^{-11}$
Yt-90	$2 \times 10^{-9}$	$5 \times 10^{-9}$	$1 \times 10^{-9}$	$1 \times 10^{-10}$	$1 \times 10^{-10}$
Zr-95	$4 \times 10^{-10}$	$1 \times 10^{-8}$	$3 \times 10^{-9}$	$3 \times 10^{-10}$	$3 \times 10^{-10}$
Nb-95	$1 \times 10^{-9}$	$4 \times 10^{-9}$	$5 \times 10^{-9}$	$3 \times 10^{-10}$	$3 \times 10^{-9}$
Mo-99	$2 \times 10^{-9}$	$2 \times 10^{-8}$	$2 \times 10^{-8}$	$1 \times 10^{-10}$	$2 \times 10^{-10}$
Ru-103	$1 \times 10^{-9}$	$5 \times 10^{-9}$	$5 \times 10^{-9}$	$3 \times 10^{-11}$	$3 \times 10^{-11}$
Ru-106	$1 \times 10^{-10}$	$6 \times 10^{-9}$	$6 \times 10^{-9}$	$3 \times 10^{-11}$	$3 \times 10^{-10}$
Cd-109	$4 \times 10^{-8}$	$1 \times 10^{-10}$	$1 \times 10^{-8}$	$6 \times 10^{-10}$	$1 \times 10^{-10}$
Cd-115m	$5 \times 10^{-9}$	$1 \times 10^{-11}$	$1 \times 10^{-9}$	$8 \times 10^{-10}$	$1 \times 10^{-10}$
Cd-115	$7 \times 10^{-9}$	$2 \times 10^{-11}$	$2 \times 10^{-9}$	$1 \times 10^{-10}$	$2 \times 10^{-10}$
Sb-124	$2 \times 10^{-10}$	$1 \times 10^{-9}$	$2 \times 10^{-9}$	$5 \times 10^{-10}$	$2 \times 10^{-10}$
Sb-125	$7 \times 10^{-10}$	$5 \times 10^{-9}$	$7 \times 10^{-9}$	$2 \times 10^{-9}$	$7 \times 10^{-10}$
I-131	$1 \times 10^{-10}$	$1 \times 10^{-10}$	$2 \times 10^{-10}$	$1 \times 10^{-11}$	$1 \times 10^{-11}$
Cs-134	$3 \times 10^{-10}$	$1 \times 10^{-9}$	$2 \times 10^{-9}$	$6 \times 10^{-10}$	$3 \times 10^{-10}$
Cs-137	$6 \times 10^{-10}$	$1 \times 10^{-9}$	$3 \times 10^{-9}$	$1 \times 10^{-9}$	$6 \times 10^{-10}$
Ce-141	$3 \times 10^{-9}$	$4 \times 10^{-9}$	$4 \times 10^{-9}$	$6 \times 10^{-10}$	$6 \times 10^{-10}$
Ce-144	$4 \times 10^{-10}$	$6 \times 10^{-10}$	$6 \times 10^{-10}$	$1 \times 10^{-10}$	$1 \times 10^{-10}$
Po-210	$5 \times 10^{-11}$	$1 \times 10^{-10}$	$1 \times 10^{-10}$	$4 \times 10^{-11}$	$4 \times 10^{-11}$
Pu-239	$2 \times 10^{-10}$	$1 \times 10^{-9}$	$5 \times 10^{-10}$	$1 \times 10^{-11}$	$1 \times 10^{-11}$

## **Radioecological Risk Assessment for Solid Radioactive Waste (SRW) Dumped into Kara Sea by the former USSR**

**V. N. Lyscov**

### **Summary:**

Sources of radioactive materials for sea disposal. Inventories of radioactivity in reactor compartments dumped with spent nuclear fuel. Inventories of induced radioactivity in reactor compartments. Characteristics of wastes containments. Problems connected with the use of "strontium equivalent" for description of SRW activity. The range of values for total SRW activity dumped into Kara Sea.

Time pattern of dumping for eight dumping sites. The assessment of radioactivity left at present for each site. Radioecological risk at present and in the future. What we need to know for planning of remedial actions.

The aim of this report is not to answer all those problems, which are mentioned in the Summary, but to identify what we positively know and what we need to know for planning of remedial actions. Other outcome, based on the balance of cost and benefits, could be complete refusal from any actions and establishment of a monitoring program providing the necessary information on the state of solid radioactive waste (SRW) dumped in the Kara Sea.

As a main database for discussion, the findings of the Russian Government Commission, chaired by A. Yablokov, will be used.

On Figure 1, all the eight areas of dumping at the inlets of the archipelago Novaya Zemlya and at the central Kara Sea depression are shown. The choice of first 6 areas was made at the beginning of sixties by the Command of the Northern Fleet. Areas 7 and 8 were chosen at a later time.

At the Table 1 coordinates and range of depths for all eight areas are given. Only Area 1 (Novaya Zemlya Depression) provides a sufficient depth and volume for a dilution of possible discharges. Other 7 areas are shallow coastal waters with great possibility of radionuclides accumulation in sediments, weeds, etc.

The choice was made due to the fact that archipelago Novaya Zemlya was practically without population and strictly guarded due to the presence of Northern nuclear test ground.

### Source Term:

The greatest concerns exist about 5 objects dumped with unloaded spent nuclear fuel (Table 2). One of these objects is the full sized submarine (length ~109 m) dumped after 13 years delay from the time of accident with two reactors on board in Stepovoy Inlet in September 1981. The others are reactor vessel, two submarine reactor compartments with two reactors each, and shielding assembly of reactor from OK-150 unit of nuclear icebreaker *Lenin*. The assessment of the activities in the objects (of the upper boundary of it) was made for the moment of dumping and it was based mainly on the total energy produced by the units. To refine these assessments, one needs the detailed information on the following:

- time and duration of reactor campaigns
- time history of power levels
- neutron energy spectra
- materials composition of the cladding and fuel assembly
- nuclear fuel composition

The present uncertainty is estimated as a factor of 2–5. The upper range of values of the total activity at the moment of dumping lays between 1000 k Ci–2300 k Ci. However, these figures could be misleading. The activity of fission products decreases approximately in such a way, if one takes the activity after 10 days delay as 1:

Time	10 days	1 month	1 year	10 years	100 years
Activity	1	0.66	0.12	0.02	0.002

Thus, after 10 years, the activity decreases 6 times in comparison with the activity after 1 year. The bulk of the dumpings of the objects with unloaded spent nuclear fuel was made about more than 20 years ago. It decreases the value of ~2 MCi to about 0.2 MCi at present. The rather recent dumping in 1981 was made after 13 years of delay. Thus our estimate of activity present today in the Kara Sea is between 0.3–0.5 MCi in fission products.

The assessments of transuranics present require more information on power levels and neutron spectra of the reactors.

The second important class of objects are those which do not contain SNF, but have high quantities of activation products formed in the reactor under the influence of neutron flux. As one can see in Table 3, only for nuclear icebreaker *Lenin* there is the assessment of induced activity in reactor vessels and reactor elements. For valid calculations, one needs to know:

- neutron fluence and neutron spectre
- duration and power level at work
- reactor geometry and geometry of other elements
- mass and materials of all the elements of assembly.

The order of magnitude of the induced activity may be assessed by simple calculation. Let us take as a reactor vessel material ferritic steel. The geometry we will model as a cylinder 0.3 x 1 x 0.14 m, covered from the inside by roast-free steel. In this case after neutron fluence  $\sim 5 \times 10^{19}$  neutrons/cm<sup>2</sup> (fission spectrum) and after 200 days of delay the activity of the object will be about 10 k Ci. We can consider it as an upper boundary estimate for activation products in reactor and surrounding reactor elements. Of course, for more reliable assessment one needs all the information which was mentioned before.

The third class of SRW is presented in Table 4. This is the most numerous and voluminous class of dumped SRW. The assessment of risk from this class of objects also requires the knowledge of source term. However, in this case it is extremely difficult due to specific method of waste activity determination. It has been done as activity of the equivalent quantity "of strontium-90" or "strontium equivalent". The idea was to use simple measurement of dose rate near container with SRW as an indicator of its total activity. It was assumed that standard container (1 x 1 x 1 m) with A = 1.0 Ci strontium-90 gives the dose rate (P) at the distance L from container P mR/h. Than for any other mixture of radionuclides its activity may be calculated by equation:

$$A = F \cdot P$$

where F is considered as a constant. This extremely crude approach, however practical it could be, leads to very crude assessments in the range of the order of magnitude. For better assessments one needs detailed description of the nature of SRW dumped.

## Barriers

For risk assessment purposes, the main interest is not in activity itself, but in what proportion it will go into the marine environment and how soon.

For the objects with spent nuclear fuel in most cases, the multibarrier protection against dissolution in sea water was used. These barriers consisted out of the walls of the construction elements and out of the fillings containing the bitumen, concrete, and a special hardening mixture on the basis of furfural. Shield assembly from icebreaker *Lenin* has been put into reinforced concrete container and metal shell. The life-time of this multibarrier protection system is estimated by the designers as something about 500 years. Only reactor from NS N421 has been put into metal container with lead shell and it was dumped together with carrying barge.

For metallic barriers several corrosion processes are possible. For austenitic rust-free steels in the environment with even low concentration of chlorine ions the rate of the growth of corrosion crackings can reach to 1 mm per day, when mechanical stress is also present. In the low stress conditions the pitting (point) corrosion is possible at the rate about 1–3 mm/a.

Other kind of material—ferritic-perlitic steel (American analogues HSLA N3,4,5, ALSI 1010) has low sensitivity to corrosion cracking. Corrosion rate in 3% NaCl solution is about 0.08 mm/a. Irradiation of this material increases corrosion rate by 2 times.

The carbon-containing steel (steel No. 3) has a corrosion rate in 3% NaCl solution between 0.07–0.5 mm/a. Irradiation enhances corrosion rate of steel 3 by 2–4 times.

The list of some Russian publications on the problems of materials corrosion in salt media under the influence of ionizing radiation is given at the end of the report.

Last two materials were widely used for construction of metallic containers for dumping of SRW of low and intermediate activity.

Taking into consideration all information presented here, and also the presence of high concentrations of dissolved oxygen in cold waters, one can assess the life-time of usual metallic containers with wall thickness 3–4 mm as 5–15 years. Thus the most part of radioactivity for low- and intermediate activity waste should be already dissolved in sea water. Some large size SRW were dumped without containers. The radioactive layer at the surface of



such objects could be between 0.1–0.5 mm. The transfer of radioactivity from the surface of such objects to sea water should also take about 5–15 years or even less. Thus, the third category of SRW could go into marine environment much more easily than the first two. At present the local contamination of the Novaya Zemlya Inlets could be mostly due to this category of SRW. The time pattern of dumping of this category of objects is illustrated by Table 4 and Figure 2.

### **Oceanography:**

The general oceanography of Kara Sea is known. The resulting transfer of water could be found in the north-east direction. However, the local characteristics of the 7 inlets of Novaya Zemlya are not known sufficiently well and it requires special field studies.

### **Ecosystems:**

The Arctic seas ecosystems have been studied long enough. Classical works by L. A. Zenkevich gave the full picture of seasonal changes in the Barentz and Kara seas (Figure 3). For Kara Sea only two seasons—spring and autumn—do exist. However, the species diversity in Kara Sea is not much less than in Barentz Sea (Figure 4). There is 51 species of fish in comparison with 104 in Barentz Sea.

But the radioecological parameters (first of all the concentration factors) for food web of Kara Sea are not well known and need to be experimentally studied.

In Figure 5, one can see the results of study by S. M. Vakulovsky and I. I. Kryshev, where dose rates for fishes from cesium-137 and strontium-90 in sea water were determined. The annual doses both in Barentz and Kara Seas are of the order  $10 \text{ } \mu\text{Gy/a}$ .

### **Ecosystems Products Consumption:**

How much of this fish will be consumed? There are practically no statistical information on the consumption of fish from the Kara Sea. The detailed information for Barentz Sea does exist.

**Doses:**

The doses for man from fish consumption from Barentz Sea were calculated by I. I. Kryshev. It was supposed that average person in Murmansk region consumes 20 kg/a and critical group—fishermen—consumes about 100 kg/a. Then the doses for the period 1960–1990 would be of the order of 10  $\mu$ Gy/a (Figure 6). The doses for man from fish consumption from Kara Sea at present can not be much more significant. The clarification of uncertainties at all stages of assessment will make the quantitative approach more reliable.

**Risk Assessment:**

The risk assessment for potential radioactive pollution of Kara Sea should include risk of human irradiation and the risk of high radioactive contamination of natural media and ecosystems. If the first component is studied and understood very well, the second component requires fresh imaginative approach. For the first component risk coefficients used by ICRP can be used. For the second, economic, socio-cultural and ecological considerations should be taken into account. For example, loss of some local endemic specie could be considered.

**Conclusions:**

The assessment of radioecological risk for SRW dumped into Kara Sea was divided into seven stages: (1) source term; (2) barriers; (3) oceanography; (4) ecosystems; (5) consumption of fish; (6) doses; and (7) risk assessment. For each stage the existing information was examined and the gaps in information necessary for quantitative solution were identified. Up to now any quantitative assessment will be highly unreliable. Accumulation of data is necessary. The crucial point is the direct study *in situ* of the dumped objects with spent nuclear fuel and with the high levels of activation products in different construction materials.

**List of:**

List of some Russian publications on the problems of materials corrosion in salt media under the influence of ionizing radiation.

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Table 1. Characteristics of Solid Radioactive Waste  
Disposal Areas in the Kara Sea

Area	Coordinates		Geographic Location	Depth, meters
	N. Lat.	E. Long.		
1	72°5' 73°17'	57°30' 60°0'	Kara Sea, Novaya Zemlya Depression	380
2	74°40' 74°42'	59°53' 60°17'	Sedov Inlet, east coast of Novaya Zemlya	13-33
3	74°35'1" 74°7'	59°18' 59°12'	Oga Inlet, east coast of Novaya Zemlya	24
4	74°22'3" 74°22'0"	58°42'1" 58°41'0"	Tsivolka Inlet, east coast of Novaya Zemlya	56-135
5	72°33'4" 72°32'4"	55°34' 55°23'3"	Stepovoy Inlet, east coast of Novaya Zemlya	25-27
6	71°56'5" 71°56'0"	55°22'1" 55°19'11"	Abrosimov Inlet, east coast of Novaya Zemlya	12-20
7	75°40'9"	63°59'	Blagopoluchiya Inlet, east coast of Novaya Zemlya	13-16
8	75°58' 75°59'	66°20' 66°18'	Techeniya Inlet, east coast of Novaya Zemlya	up to 50

Table 2 Objects with Spent Nuclear Fuel Dumped in Northern Seas

Object	Coordinates, Year	Depth, meters	Total Activity (max.), kCi*	Radionuclide Content	Description of Protective Barriers
Compartment of NS No. 285 with two reactors, one containing SNF in place (see also Table 4)	71°56'2" N, 55°18'5" E, Abrosimov Inlet, 1965	20	800	Fission products	Stock reactor compartment and interior structures filled with furfural mixture
Compartment with two reactors containing SNF from NS No. 901	71°56'2" N, 55°18'9" E, Abrosimov Inlet, 1965	20	400	Fission products	Same
Shielding assembly of reactor from OK-150 unit of nuclear icebreaker <i>Lenin</i> with residual SNF (60% of fuel complement based on $UO_2$ )	74°22'1" N, 58°42'2" E, Tsivolka Inlet, 1967	49	100	$^{137}Cs$ (~50 kCi), $^{90}Sr$ (~50 kCi), $^{238}Pu$ , $^{241}Am$ , $^{244}Cm$ (~2 kCi)	SNF residue bound by furfural-based mixture, shielding placed in reinforced concrete container and metal shell
Reactor from NS No. 421 with SNF	72°40' N, 58°10' E, Novaya Zemlya Depression, 1972	300	800	Fission products	Metal container with lead shell dumped along with barge
NS No. 601 with two reactors containing SNF	72°31'15" N, 55°30'15" E, Stepovoy Inlet, 1981	50	200	Fission products	Stock reactor compartment and interior structures filled with furfural mixture
Total: 5 objects with 7 reactors containing SNF	1965-1981		2300		

\*—Expert estimates were made at the time of sinking, based on power generated by NS reactors (12.5 GW/day).

Table 3 Objects without Spent Nuclear Fuel Dumped in Northern Seas, 1965-1968

Object	Coordinates, Year	Depth, meters	Total Activity	Radio- nuclide Content	Description of Protective Barriers
Reactor of NS No. 285 (see Table 3)	71°56'2" N, 55°18'5" E, Abrosimov Inlet, 1965	20	Requires special analysis	Unclear	Stock reactor compartment structures
Reactor compartment (two re- actors) from NS No. 254	71°55'13" N, 55°32'32" E, Abrosimov Inlet, 1965	20	Requires special analysis	Unclear	Stock reactor compartment structures
Reactor compartment (two re- actors) from NS No. 260	71°56'2" N, 55°18'5" E, Abrosimov Inlet, 1966	20	Requires special analysis	Unclear	Stock reactor compartment structures
OK-150 nuclear power plant from icebreaker <i>Lenin</i> , compri- sing three reactors with prima- ry loop pipelines and water- tight stock equipment	74°26'4" N, 58°37'3" E, Tsivolka Inlet, 1967	50	~50 kCi	Mainly <sup>60</sup> Co	Biological shielding unit (B-300 steel, concrete)
Two reactors from NS No. 538	73°59' N, 66°18' E, Teheniya Inlet, 1988	35-40	Requires special analysis	Unclear	Metal con- tainer with lead shell
Total: 5 objects with 10 reac- tors without SNF	1965-1988	20-40	Requires special analysis (possibly up to 100 kCi at time of dunning)		*

Table 4. Characteristics of Solid Radioactive Waste Dumping in Northern Seas

Year	Coordinates		Volume, m <sup>3</sup>	Activity ( <sup>90</sup> Sr equivalent, Ci)	Form of Disposal		
	N. Lat.	E. Long.			Containers	Ships	Unenclosed
Area 1. Kara Sea							
1967	73°17'3"	59°54'	212	35.3			Main circulating pump from nuclear icebreaker <i>Lenin</i> (3 pcs.)
1967	72°21'	57°50'18"	910	359		Steamer <i>José Diaz</i>	
1968	73°06'	59°10'	150	5.6		Barge No. 3	
1969	Area 1		144.8	159.2	?	?	?
1970	73°11'	59°54'	144	5.6	?	?	?
1972	72°24'	57°55'	?	160		Lighter <i>Sayany</i>	
1973	72°23'	58°0'	?	?		Tanker <i>TNT15</i>	
1974	72°11'	57°40'	?	?		Lighter <i>Oma</i>	
1975	72°38'	58°20'	5000	30		Lighter <i>L-3</i>	
1977	72°19'22"	57°46'	600	0.6		<i>MBSN-801250</i>	
1980	72°18'1"	57°36'4"	243	118.4			
1980	72°15'	57°35'	?	?			
1984	72°15'	57°30'	295.1	248.6	Containers (V+)		.
1984			4.0	5.8			Class III furfural-acetone resin (V+)
1984			3.0	14.8			Primary loop circulating pump
1985	72°21'	57°50'18"	5182.1	738.24	1027		SRW
1985	73°06'	59°10'	693.26	506.99	535		
1986	72°21'	57°50'18"	419.4	156.83	321		
1987	73°06'	59°10'	1302.3	628.14	847		Steam generator, primary loop circulating pump
1989	73°06'	59°10'	370.26	87.095	256		
1989	72°21'	57°50'18"	142	24.18	57		
1991	73°17'3"	59°54'	264.4 54.5	20.66 14.92	131		SRW
Total			16134	3320 (123)	3174	8	9

Table 4 (continued)

Year	Coordinates		Volume, m <sup>3</sup>	Activity ( <sup>90</sup> Sr equivalent, Ci)	Form of Disposal		
	N. Lat.	E. Long.			Containers	Ships	Unenclosed
Area 2. Sedov Inlet							
1982	74°40'	59°55'	?	100			
1982	74°40'	59°55'	2357.6	1718.2	298		91 4k-650B bundles
1982	74°42'	69°56'	218.4	63.56	182		
1982	74°41'	59°53'	276	118.32	230		
1983	74°40'	59°56'	280.5	1121.44	231		
1984	74°41'	60°17'	136.5	172.8	108		
1984	74°41'	60°17'	3.0	6.0			Furfurol- acetone resin (6 pcs.)
1984	74°41'	60°17'	10.5	52.5			Primary loop circulating pump
1984	74°41'	60°17'	150.9	57.21	59		Steam gene- rator
Total			3433	3140 (126)	1108		104
Area 3. Oga Inlet							
1968	74°07'	53°12'	400	4		Barge <i>SB-5</i>	SRW
1976	74°35'1"	59°15'4"	560	929			
1978	74°17'	58°18'	170	15.5			SRW
1980	74°35'	59°14'	278	274.35			SRW
1980	74°35'	59°14'	500	59.21			*
1981	?	?	?	349.06	containers, ?		SRW
1983	74°35'	59°13'5	540	205.32	212		
1983	74°35'1"	59°13'1"	580	190.6	260		
Total			3028	2027 (75)	472+?	1	4
Area 4. Tsiivka Inlet							
1964	74°22'3"	58°41'	640	977.37	1600	Special lighter <i>N. Bauman</i>	SRW
1965	74°22'3"	58°41'	266	448.96			SRW
1966	74°22'3"	58°41'	446	534.17			SRW
1967	74°22'3"	58°42'	240	374.97			SRW
1967	74°22'2"	58°41'5"	25.2	28.64			SRW
1967	74°22'3"	58°42'1"	72.2	77.2			SRW
1976	74°22'	58°42'	1233	12		Special lighter <i>Kolezhma</i>	
1978	74°22'	58°41'	438	230.5	?		
Total			3360	2684 (99)	1600+?	2	6



Table 4 (continued)

Year	Coordinates		Volume, m <sup>3</sup>	Activity ( <sup>90</sup> Sr equivalent, Ci)	Form of Disposal		
	N. Lat.	E. Long.			Containers	Ships	Unenclosed
Area 5. Stepovoy Inlet							
1968	72°32'4"	55°33'9"	185.2	184.78			SRW from nuclear ice- breaker <i>Le- nin</i>
1970	72°33'	55°29'2"	243	371.12			SRW
1972	72°33'2"	55°26'2"	242	212			SRW
1973	72°33'2"	55°23'3"	532	325.24			SRW
1975	72°33'4"	55°24'	445	187			SRW
Total			1647	1280 (47)			5
Area 6. Abrosimov Inlet							
1966	71°56'1"	55°19'5"	?	?		Barge	
1967	71°56'5"	55°21'5"	?	0.28		Barge <i>MNN- 231500</i>	
1967	71°56'	55°21'	?	30		Barge <i>MBSN- 378250</i>	SRW
1974	71°56'0"	55°21'0"	520	229			
1977	71°55'3"	55°22'1"	254.8	387	8		
1980	71°56'	55°21'	750	10		Lighter <i>L-8711</i>	Steam gene- rators (5 pcs.), SRW
1981	71°56'	55°21'2"	392	5	?		
Total			1917	661 (24.5)	8+?	4	7
Area 7. Blagopoluchiy Inlet							
1972	75°40'9"	63°39'	331	234.84			SRW from <i>Lenin</i>
Total			331	235 (8)			1
Area 8. Techeniy Inlet							
1982	76°58'	66°20'	91.2	29.34	76		
	"	"	84.0	4.0	70		
1988	73°59'	66°18'	229	1811.21		Lighter No. 4	
Total			404	1845 (68)	146	1	

Table 4 (continued)

Year	Coordinates		Volume, m <sup>3</sup>	Activity ( <sup>90</sup> Sr equivalent, Ci)	Form of Disposal		
	N. Lat.	E. Long.			Containers	Ships	Unenclosed
Outside Areas 1-8							
1978	69°34'1"	47°56'3"*	1100	40		Lighter <i>Nikel</i>	<i>SRW</i>
Total			1100	40 (1.5)		1	18
Grand Total			31534	15502 (574) TB <sub>γ</sub>	6508	17	154

\*—20 miles northwest of Kolguyev Island, possible coordinates 69°34'0" N, 47°56'2" E.

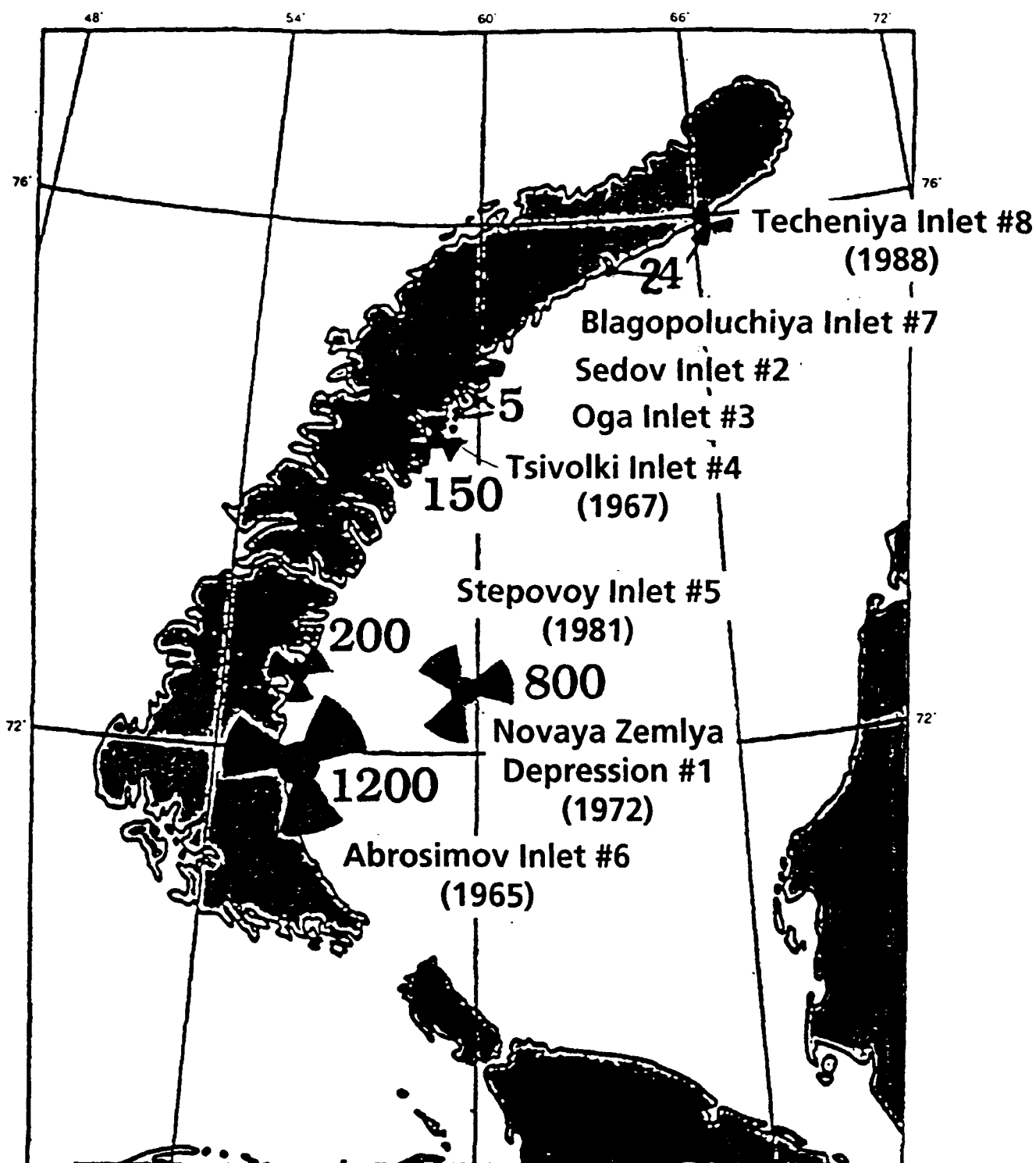


Figure 1. Expert Estimates of Maximum Possible Total Activity (at time of disposal) of All Forms of Solid Radioactive Waste in the Kara Sea.

Size of symbol is proportional to activity; figures are in kCi. Exact coordinates of areas given in Tables 1 and 4.

$A_{410}^{Ci}$

АКТИВНОСТЬ, Ки

2000

$Ci$

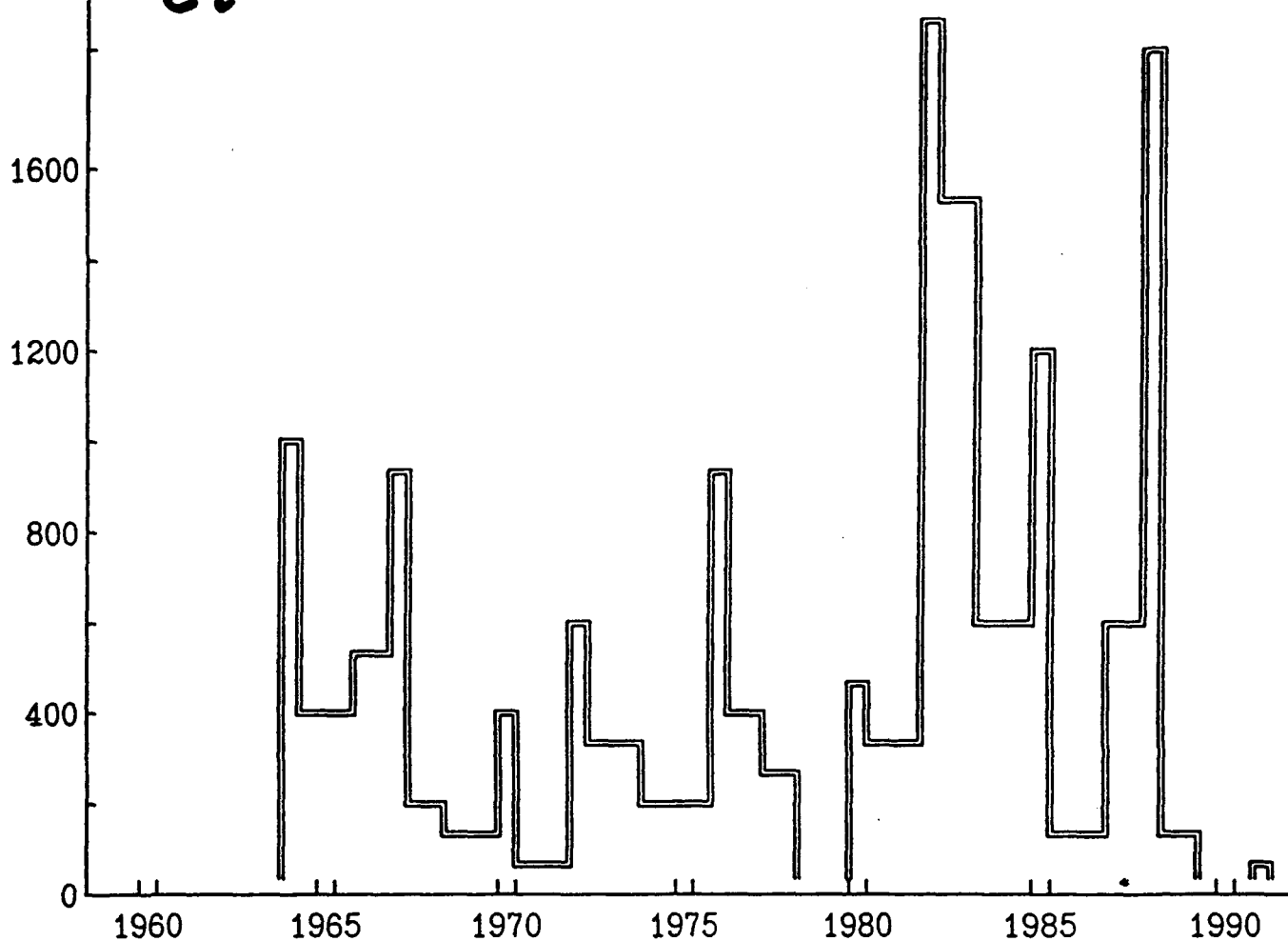


Рис. 3.2 Сброс ТРО в северные моря (по данным табл. 3.4а)

years

Примечание. Без учета захоронений, указанных в табл. 3.2, 3.3.

Fig.2 SRW dumping into Kara sea according to the data of Table 4.

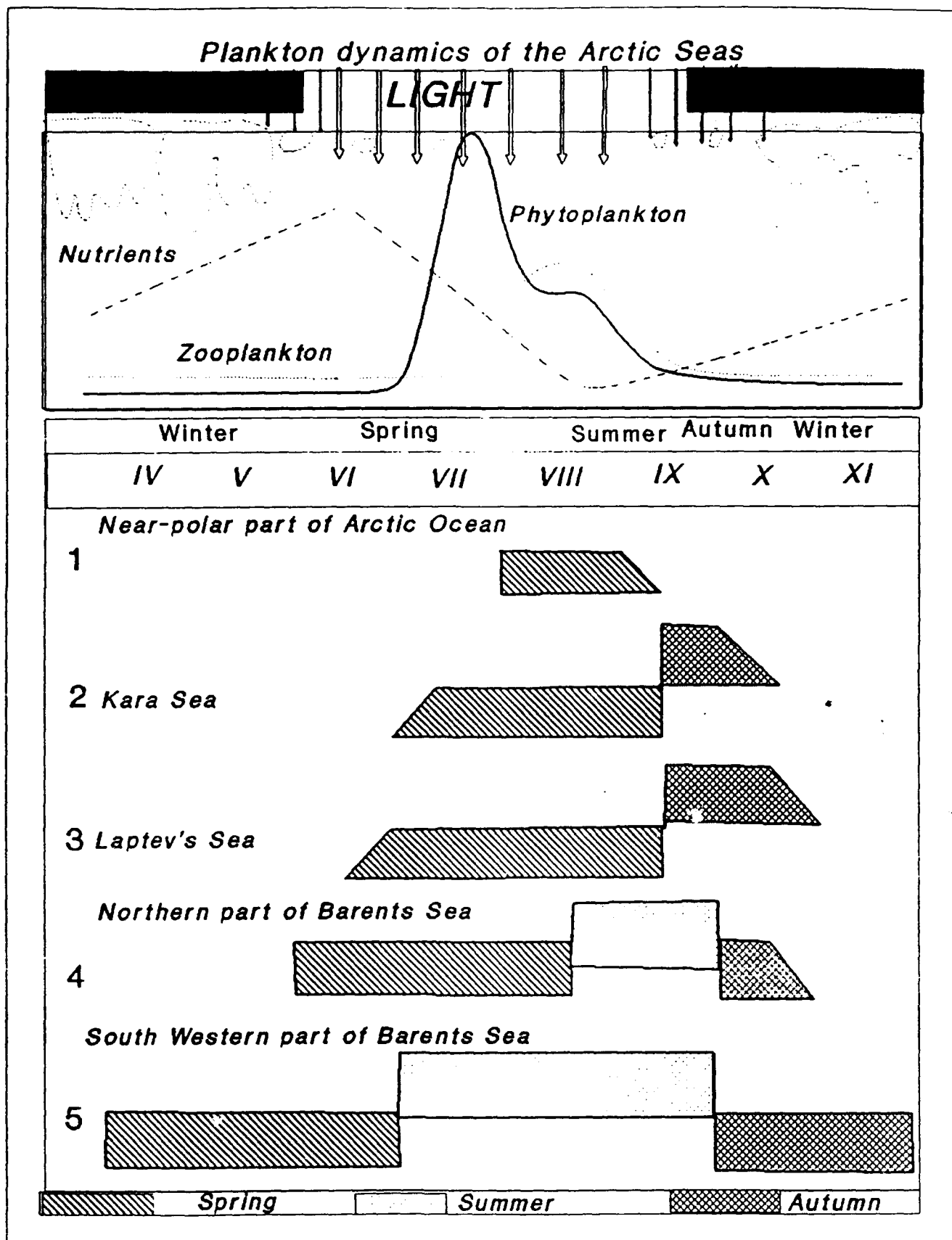


Fig 3

according to A. Zenkevich.

# Species Diversity of the Arctic Seas

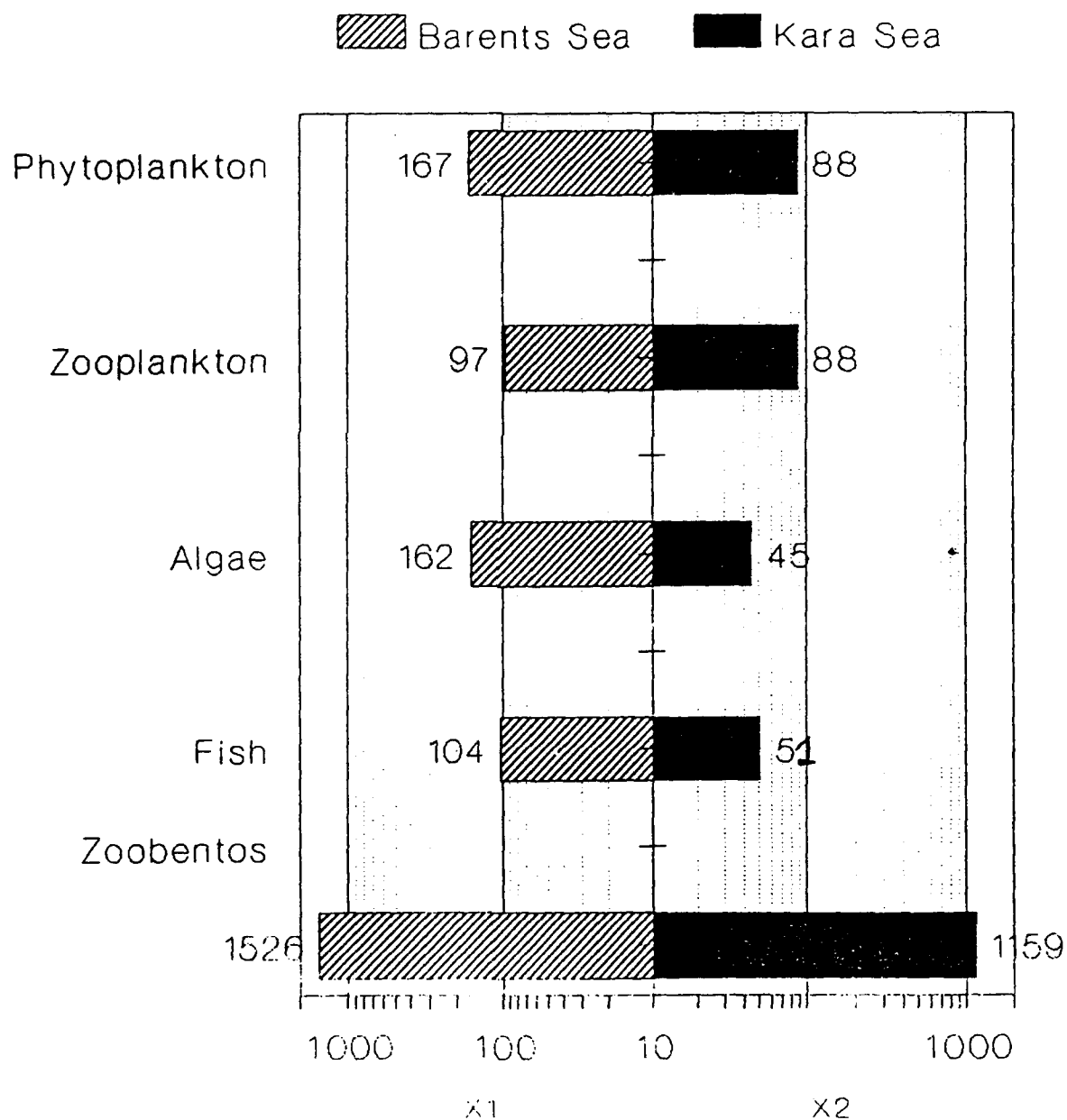
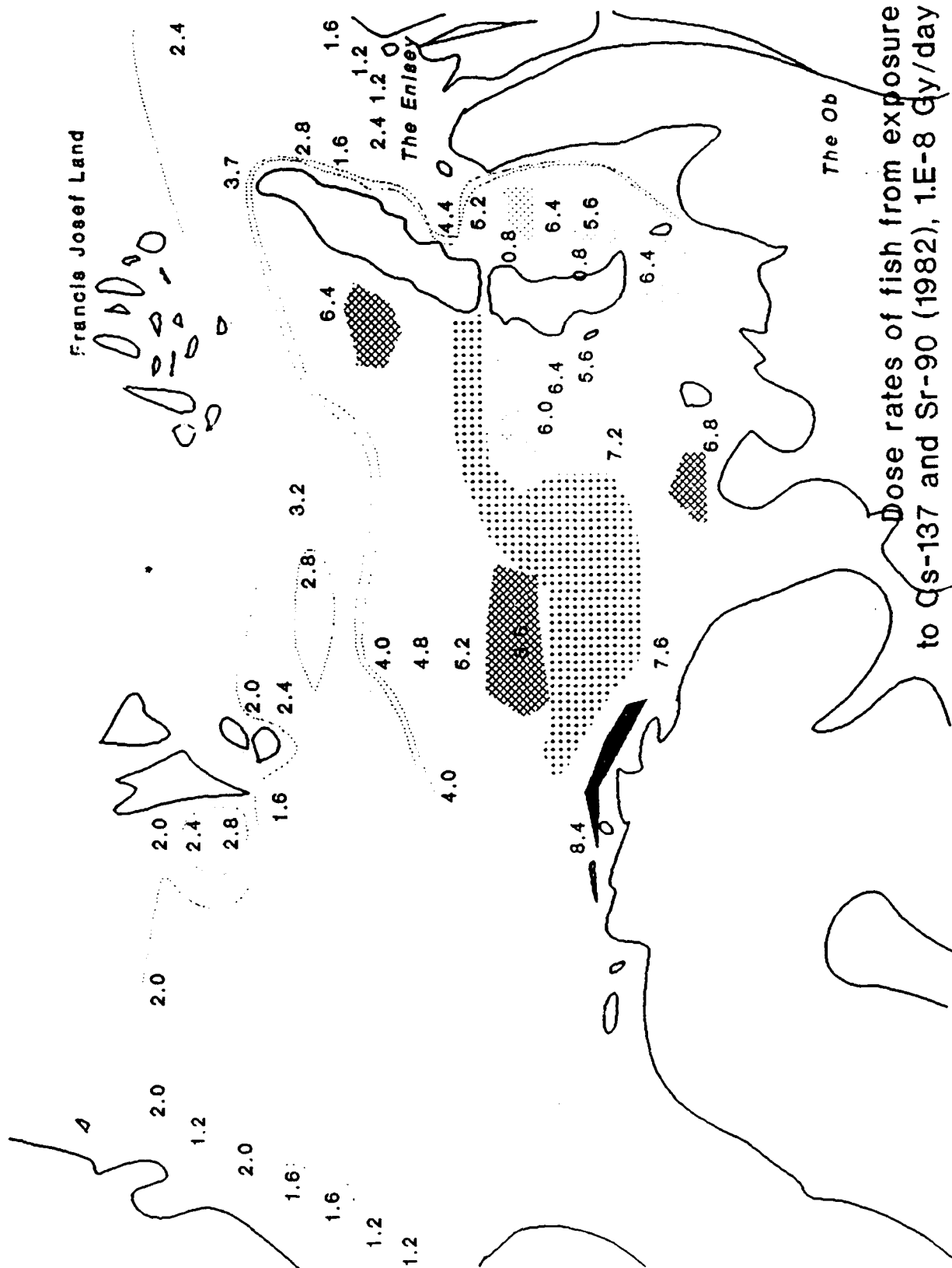


Fig 4

according to A Zenkevich



Dose rates of fish from exposure  
to Cs-137 and Sr-90 (1982), 1.E-8 Gy/day

according to S. M. Vakulovsky  
I. I. Kryshev

Fig 5

# Dose reconstruction from consumption fish in the Barents Sea

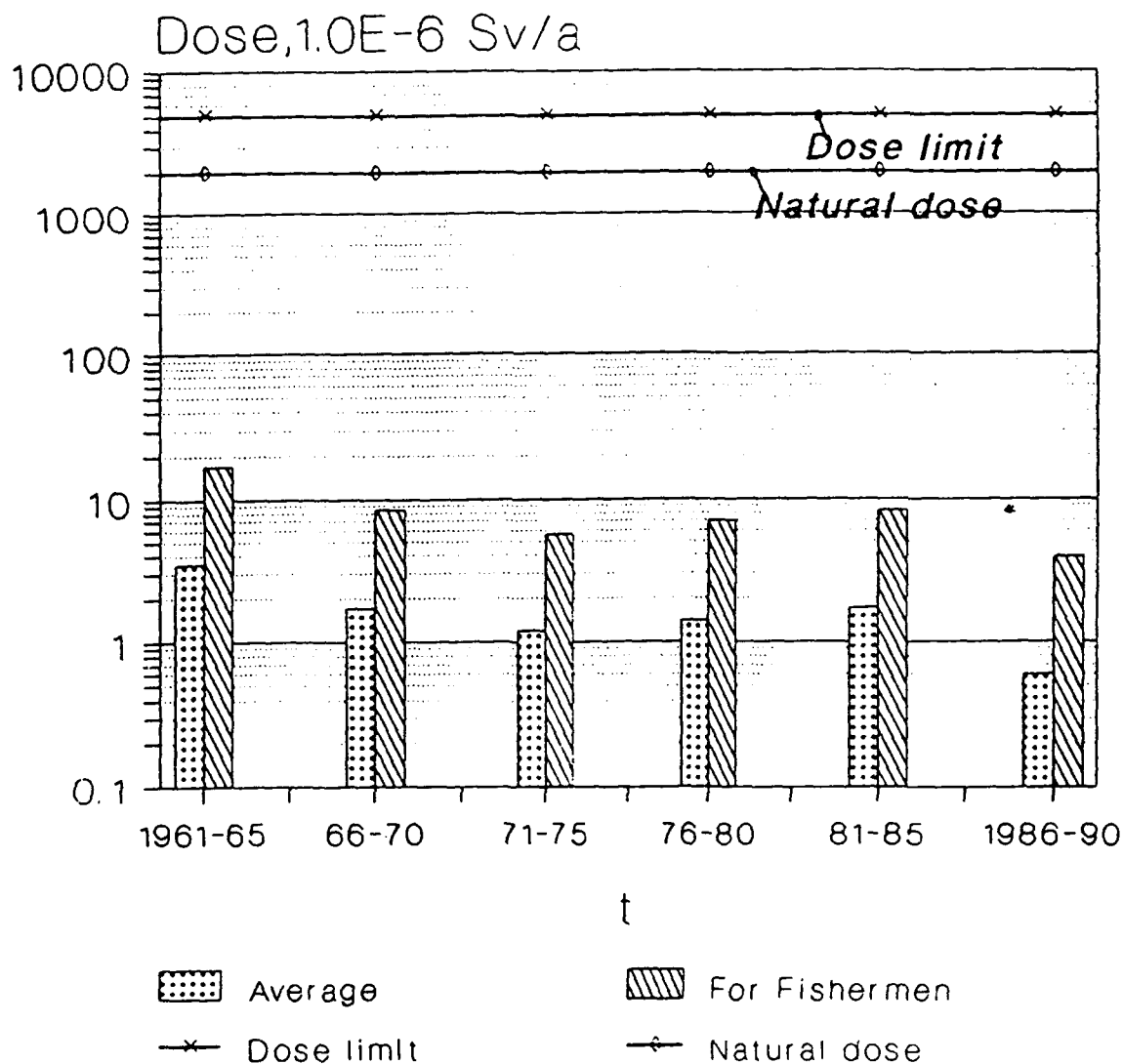


Fig 6

according to I. I. Kryshev



## Radiological Assessment of the Ribble Estuary

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### Abstract

*The Ribble Estuary, UK receives radionuclides derived from both BNFL Sellafield and Springfields. The beta and gamma dose rates have been assessed at 87 sites within the estuary together with the surface sediment radionuclide activity concentrations. Habit surveys have identified the groups and individuals who use the Ribble Estuary for recreation and work and their annual dose, based on their occupancy, has been assessed. The maximum assessed dose was 73  $\mu$ Sv for walkers on Lytham Marshes although this is only half the critical group (houseboat dwellers) identified by MAFF. Of the total doses, the inhalation and beta emitting radionuclides are relatively small components, up to 16.5%. The majority of the total dose is derived from Sellafield-derived gamma emitting radionuclides and not the high activity beta discharges from Springfields. The dynamic nature of the fine-grained sediments within the estuary is also indicated.*

### Introduction

British Nuclear Fuels plc (BNFL) operate a uranium fuel fabrication plant at Springfields near Preston, U.K. The company are authorised to discharge radionuclides arising from the purification of uranium ore concentrates and reprocessed uranium. These radionuclides include the short-lived beta emitters of the  $^{238}\text{U}$  decay chain,  $^{234}\text{Th}$  and  $^{234\text{m}}\text{Pa}$ . These discharges are made through a pipeline to the Ribble Estuary, arising approximately 20 km from the mouth of the estuary and 11 km from the tidal limit. Continuous discharges are made although the radionuclide content varies greatly depending on the amount of fuel being fabricated.

BNFL also operate a spent nuclear fuel reprocessing facility at Sellafield, west Cumbria, U.K. The reclamation of uranium from the fuel gives rise to fission and activation products, a proportion of which are discharged under authorisation to the Irish Sea. The principal radionuclides of importance in these discharges are relatively long-lived alpha and gamma emitters such as  $^{137}\text{Cs}$ ,  $^{241}\text{Am}$  and Pu

isotopes. Discharges, whilst not constant, are made continuously over the year. Through the action of tidal currents in the Irish Sea, some of these radionuclides can be found in the Ribble Estuary.

The radionuclides present in the estuary will lead to a dose to the public using the Ribble Estuary for recreation or work. This paper presents the results of a radiological assessment of the Ribble Estuary and indicates the relative importance of each source of radionuclides.

### Beta emitting radionuclides and dose rates

Beta dose rates were measured 30 cm above the sediment at 87 sites around the Ribble Estuary (Figure 1). Four separate visits were made using a Berthold LB1210B measuring instrument. Details of the methodology used can be found in Mudge *et al.*, (a. in submission).

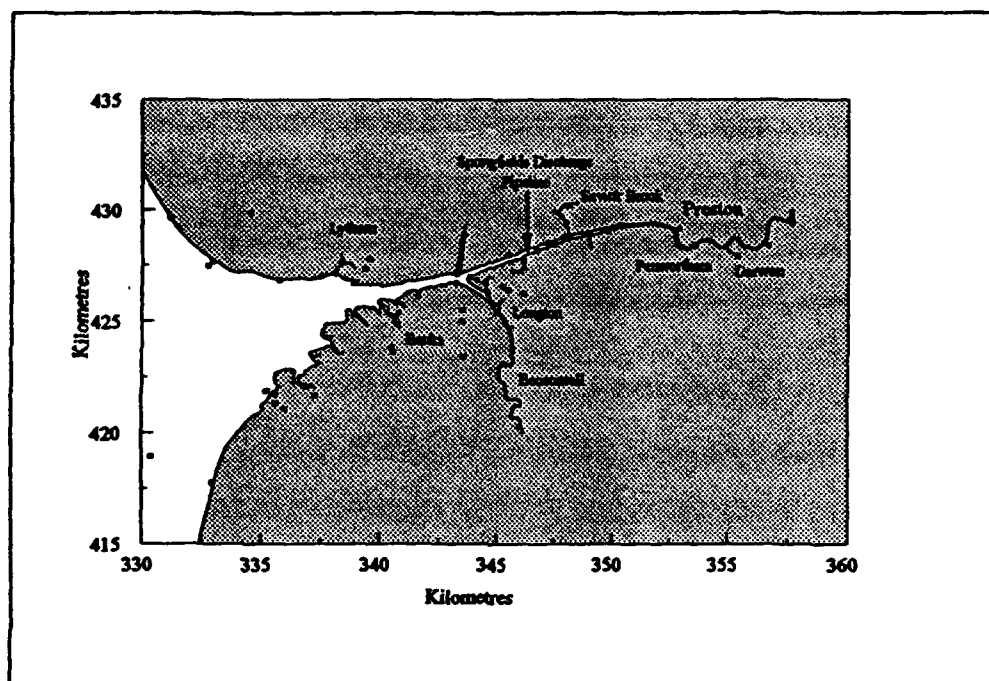


Figure 1. Map of the Ribble Estuary showing the location of sampling sites.

During the four surveys, beta dose rates were seen to increase to a maximum ( $\sim 20 \mu\text{Sv}\cdot\text{h}^{-1}$ ) in January, 1992 (Figure 2(a)). The higher beta dose rates were mainly associated with deposits of fine-grained sediments in the upper reaches of the estuary. The distribution of  $^{234\text{m}}\text{Pa}$  activity in surface sediments at this time can be seen in Figure 2(b). There is a statistically significant correlation between the beta dose rate and the surface sediment  $^{234\text{m}}\text{Pa}$  distribution. Between January and April, 1992 increased river flow led to a redistribution of the fine-grained sediments. There was a net transport of fine-grained sediments from the inner estuary to the outer estuary, *e.g.* Lytham, producing a smoothing of the previously localised high activities in sediments of the inner estuary.

After discharges from Springfields ceased in October, 1992, further visits were made to assess the environmental decay characteristics of the radionuclides. The surface activity concentrations of  $^{234}\text{Th}$  at three sites are presented in Figure 3 together with their theoretical decay curves. Three different decay patterns can be seen.

1. At the discharge pipe, the  $^{234}\text{Th}$  activity concentration declines in close agreement with the theoretical decay curve (Fig. 3(a)). This suggests that there is little net sediment movement to or from the site.
2. In Savick Brook (Fig. 3(b)), the activity concentration decreases significantly quicker than radioactive decay would predict. This is due to the erosion of contaminated fine-grained sediments from this site. Some of the deposits in the upper reaches of these tributaries are thin and overlie agricultural soils. They are deposited on high spring tides and can be removed by increased river flow or subsequent high tides.
3. At Beconsall (Fig. 3(c)) the activity declines slower than the decay curve predicts. This is due to inputs of contaminated sediments derived from other sites within the estuary such as Savick Brook and not fresh sediment from the Irish Sea which would only be contaminated with the Sellafield derived radionuclides.

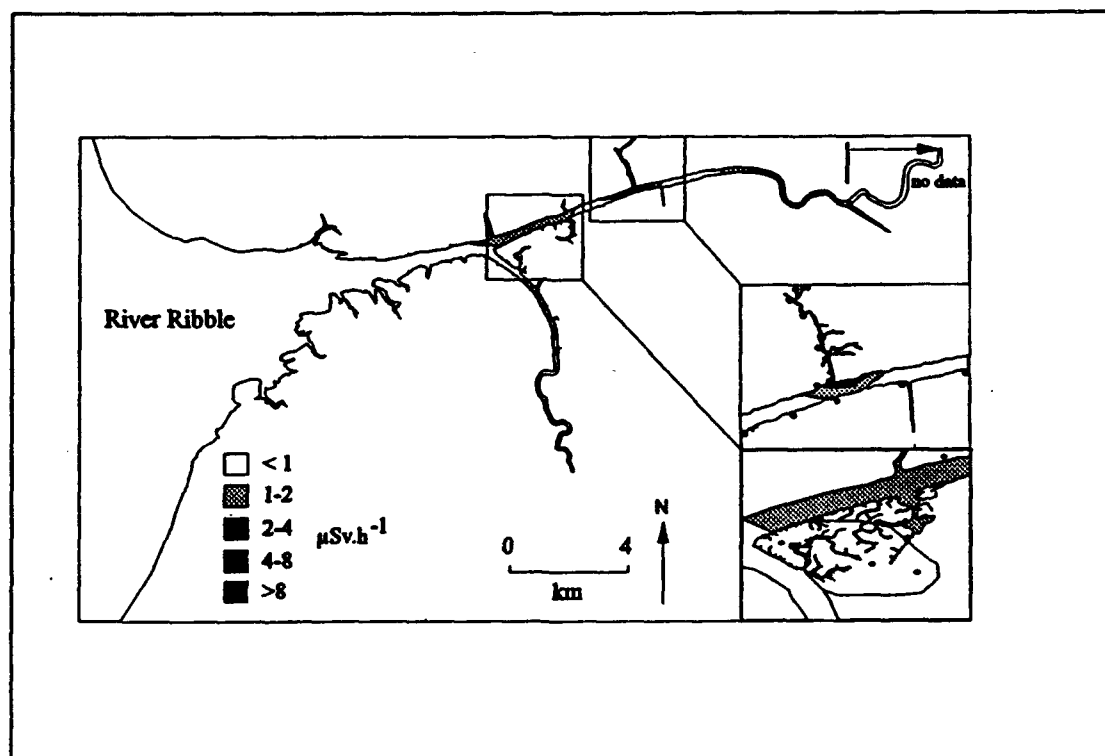


Figure 2. (a) Beta dose rate ( $\mu\text{Sv.h}^{-1}$ ). Measurements were made in January, 1992.

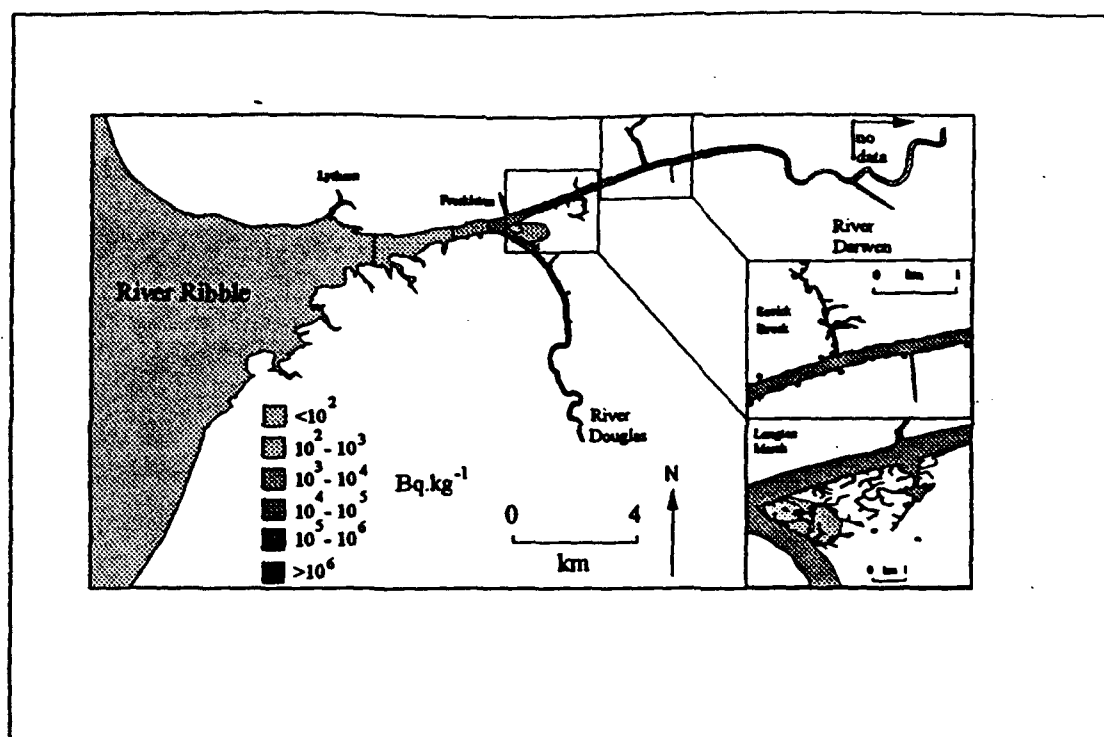


Figure 2. (b) Distribution of  $^{234m}\text{Pa}$  in the surface sediments ( $\text{Bq.kg}^{-1}$ )

### Gamma emitting radionuclides and dose rates

Gamma dose rates were measured at 1 m above the sediments using a Mini-Instruments 6-80 with a gamma compensated G-M tube held vertically (see Mudge *et al.* a. in submission for further details). The gamma dose rate is principally derived from Sellafield discharged radionuclides although correlation with individual radionuclides is less pronounced compared to  $^{234m}\text{Pa}$  and the beta dose rate. Gamma dose rates were measured at the same sites as those in the beta dose rate survey although there was little variation between surveys (Fig. 4(a)). The maximum dose rate ( $192 \text{ nGy.h}^{-1}$ ) was seen on the raised salt marshes of Longton. These are old, thick deposits containing sediments contaminated with past Sellafield discharges when activity concentrations were higher. This, in part, explains the relatively poor correlation between the gamma dose rate and the surface activity concentrations. A better correlation may be found if the radionuclide inventory to 30 cm was used instead. The highest surface sediment activity concentrations for  $^{241}\text{Am}$  (up to  $522 \text{ Bq.kg}^{-1}$ ) were also associated with fine-grained sediments (Fig. 4(b)) although the spatial distribution is somewhat different to that of  $^{234}\text{Th}$  due to the source being the marine end of the estuary.

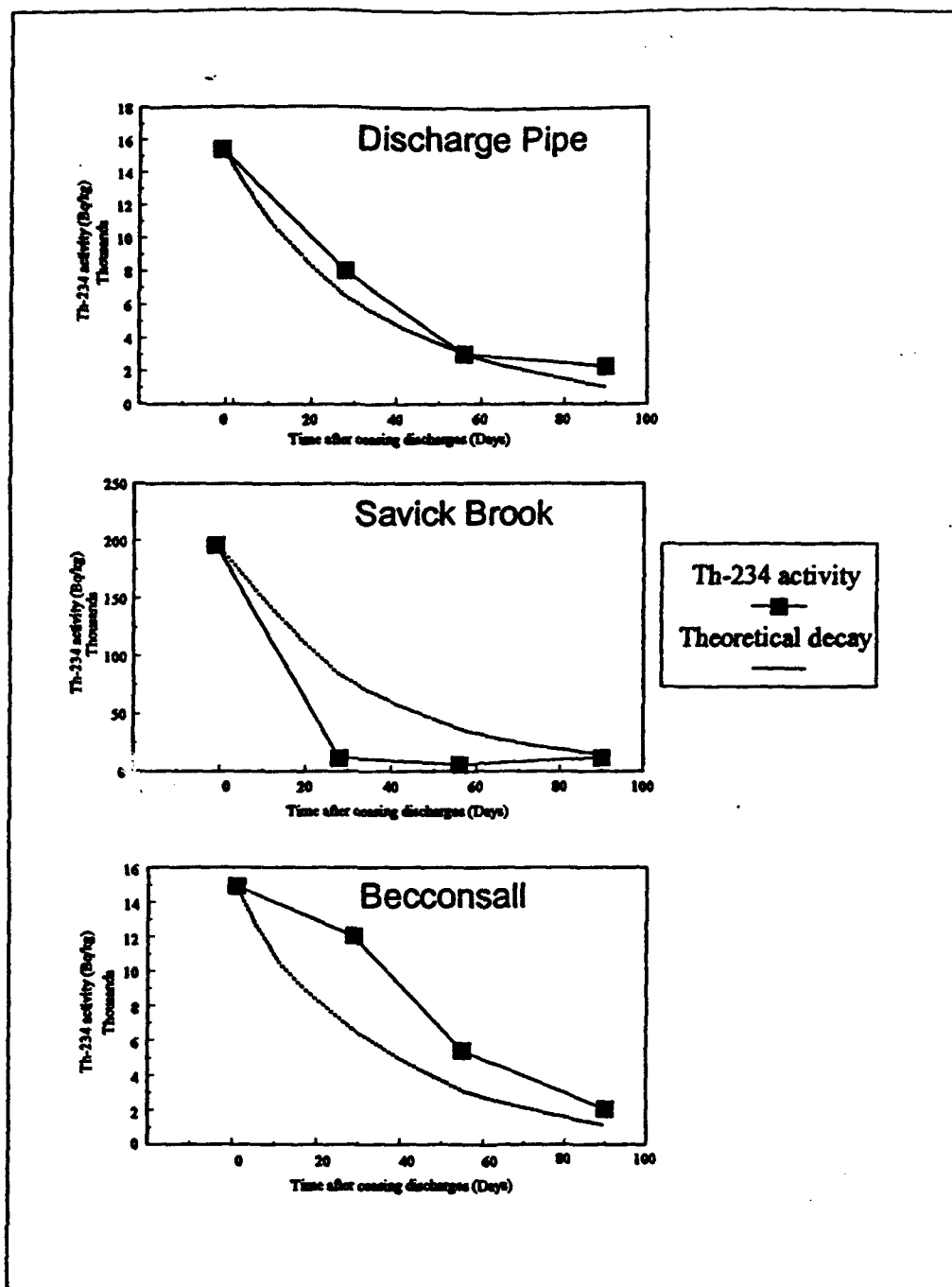


Figure 3. Change in surface sediment  $^{234}\text{Th}$  activity concentration at (a) the discharge pipe, (b) in Savick Brook and (c) at Becconsall.

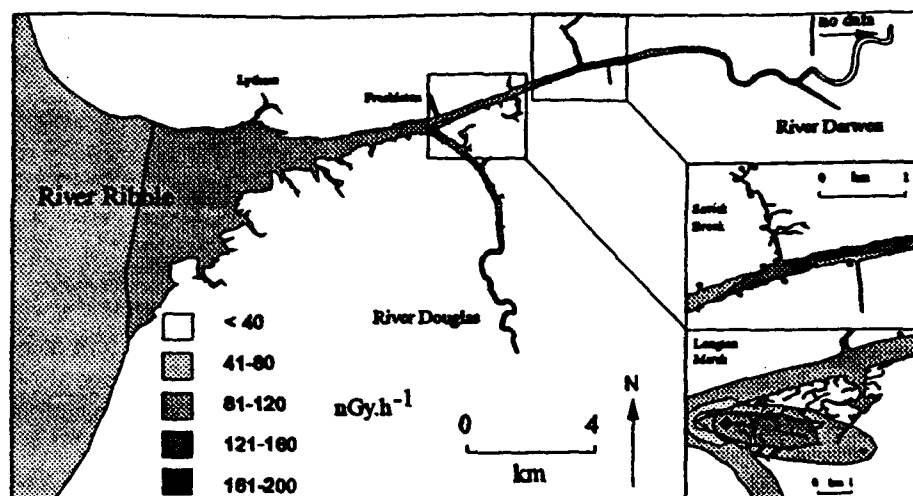


Figure 4. (b) Typical gamma dose rate ( $\text{nGy.h}^{-1}$ ). Measurements were made in January, 1992.

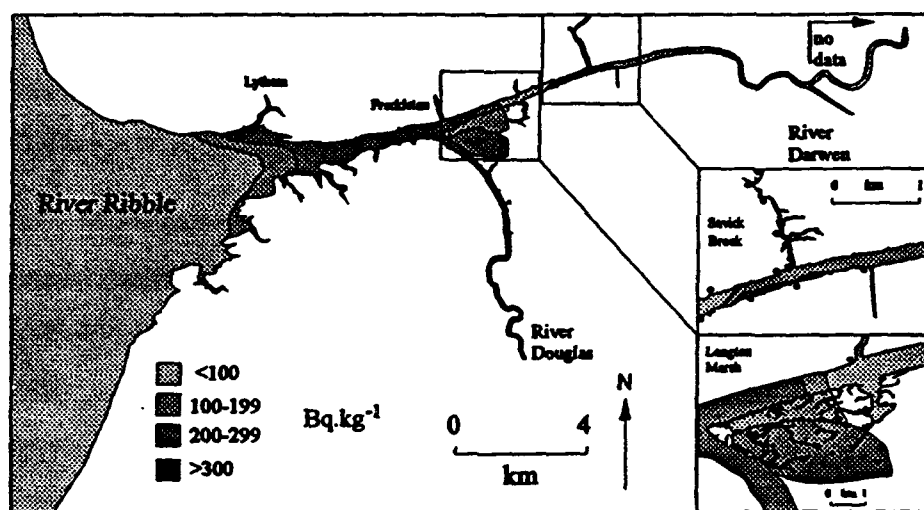


Figure 4. (b) Distribution of Am-241 in surface sediments ( $\text{Bq.kg}^{-1}$ )

After discharges from Springfields ceased, similar decay behaviour to that of the  $^{234}\text{Th}$  could be seen for the  $^{137}\text{Cs}$  (Fig. 5). At the discharge pipe (Fig. 5(a)), no significant changes in  $^{137}\text{Cs}$  activity concentrations could be seen. In Savick Brook (Fig. 5(b)) the surface sediment activity concentration declined rapidly over a period of 90 days whereas radioactive decay would suggest no perceptible changes. This confirms the above results of significant sediment removal and relocation at this site. At Becconsall (Fig. 5(c)) the activity concentration increased over the 90 day period. This is in response to fresh sediment from the Irish Sea and the redistributed sediments from other sites about the estuary.

## Dose Assessment

The beta and gamma dose rates measured above can only lead to a human dose if people visit these areas. To assess this aspect of the Ribble Estuary, a number of techniques were employed (see Table 1 and for a fuller treatment see Mudge *et al.*, b. in submission).

Table 1. Methods and effectiveness used in occupancy assessment.

Method	Success rate
Face-to-face questionnaires	196 on 41 visits
Posted questionnaires to clubs	115 sent with 40% replying
Time-lapse video photography	15 days

From the above data, the annual occupancies of individuals and groups were estimated. A number of dose pathways were considered including external beta and gamma irradiation, irradiation of the gonads and eyes and inhalation. The full results are presented elsewhere (Mudge *et al.*, 1993) but the principal aspects are shown below (Table 2).

The maximum dose assessed was 73.3  $\mu\text{Sv}$  for walkers with a high occupancy on Lytham Marshes. The other doses are less than half of this value. Anglers receive a dose from standing and sitting on the banks near the tidal limits of the estuary. Wildfowlers lie on the sediments and receive doses up to 36.2  $\mu\text{Sv}$ . Walkers receive doses dependent on where they go; whilst the dose rates are relatively high in Savick Brook, people rarely go there. The inhalation pathway was assessed by measurement (high volume air sampling) and modelling. In both cases, the significance of this pathway appears to be low (Mudge *et al.*, 1993). Doses to individual organs (eyes and gonads) are less than 10% of the non-stochastic dose limit suggested by ICRP60 (Phipps *et al.*, 1991). The critical group for the Ribble Estuary has been identified by MAFF (Camplin, 1993) as the houseboat dwellers who live up the muddy creeks of the river. Their dose for 1991 has been assessed to be 150  $\mu\text{Sv}$ , twice that of the non-resident users of the estuary.

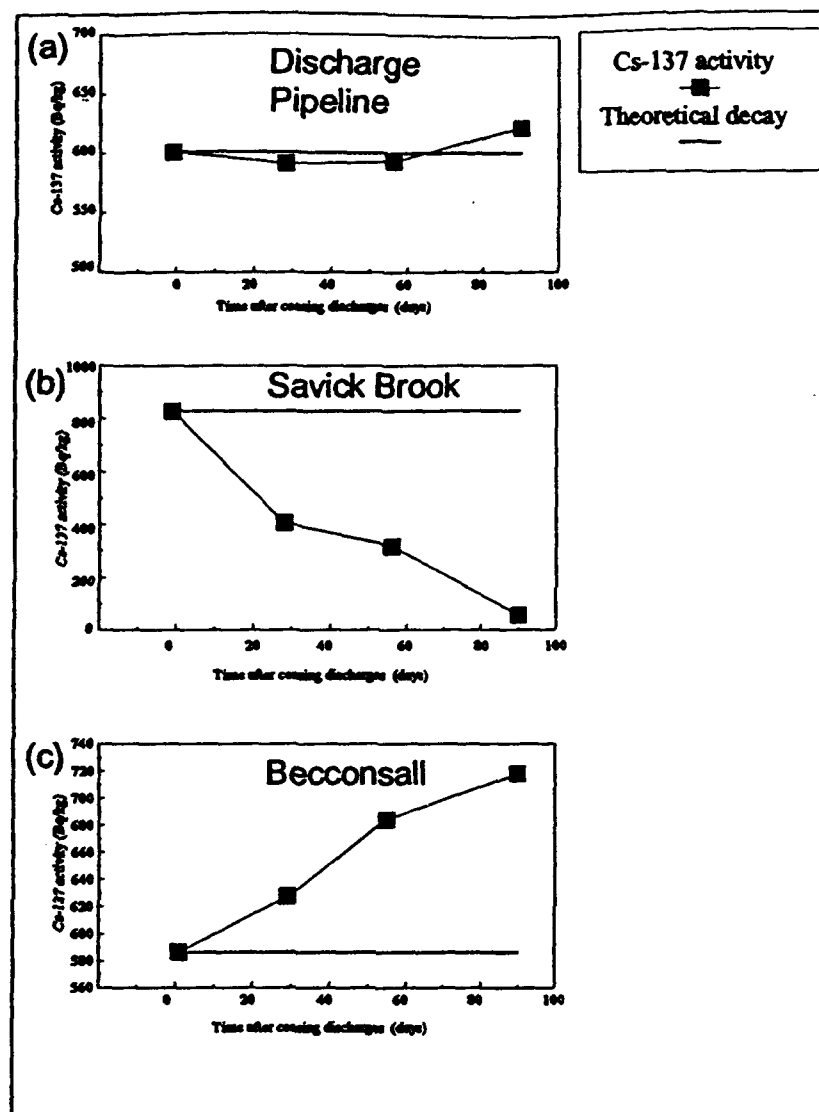


Figure 5. Change in surface sediment  $^{137}\text{Cs}$  activity concentration at (a) the discharge pipe, (b) in Savick Brook and (c) at Beconsall.

Of the total dose, the majority is derived from the gamma emitting radionuclides of Sellafield origin. Even though the surface sediment activity concentrations for  $^{234\text{m}}\text{Pa}$  can exceed  $1 \text{ MBq.kg}^{-1}$ , the maximum beta dose component of the total dose is 16.5% (Table 2). Therefore, it is the artificial radionuclides discharged from Sellafield into the marine environment some 80 km distant that produce the largest increase in public dose in the Ribble Estuary. The previously high discharge rates of these radionuclides (BNFL, 1979-1992) coupled with their long half lives, will lead to elevated doses for the foreseeable future.



Table 2. Dose assessment for selected users of the Ribble Estuary

Users	Sites	Gamma ( $\mu\text{Sv}$ )	Beta ( $\mu\text{Sv}$ )	Inhalation ( $\mu\text{Sv}$ )	TOTAL ( $\mu\text{Sv}$ )
Anglers	Penwortham	14.1	5.9	0.8	20.8
Wildfowlers	Banks	15.3	1.7	0.3	17.3
	Longton	34.1	1.7	0.4	36.2
	Lytham	30.9	3.9	0.4	35.2
Walkers	Penwortham	22.1	2.8	2	26.9
	Lytham	60.6	10	2.7	73.3
	Savick	6.5	0.7	0.1	7.3

### Acknowledgements

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**PROBLEMS OF EVALUATION AND MANAGEMENT OF  
RISK OF RADIOACTIVE CONTAMINATION OF OCEANS.**

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**Radioactivity and Environmental Security in the Oceans:  
New Research and Policy Priorities in the Arctic and  
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We consider in this paper the following three trends in Risk Analysis and Management, which are essential for the problem of radioactive contamination of oceans.

- Expert analysis of factual risk from antropogenic radioactive contamination of oceans and comparison of this risk with other risks of radioactivity in oceans.

- Regional aspects of risks associated with transboundary transfers of radioactive substances.

- Short time goals and directions of further research.

### 1. Global Contamination of Oceans.

Not focusing on the problem of radioactive waste deposits and International Law, here we note that the proposal of immediate moratorium on radioactive dumps have not been supported by all countries which deposit low- and medium-active substances in the oceans. Some of the countries doubted the extent of hazard, which this practice poses to the humanity. At the same time the governments are aware of economic difficulties which would arise if the proposed moratorium be adopted. The analysis of risk of radioactive dump sites in the oceans, performed by IAEA [1] was discussed at the Fourth IGPRAD Conference. It was criticized for large incertainties involved in assessment of risk of these hazards.

What is the source of these uncertainties? According to the latest Russian report on radioactive contamination of the oceans [2], the upper limit of total radioactive contamination of the oceans is evaluated as  $5 \cdot 10^{17}$  Bk. Being totally dissolved in the ocean water, antropogenic radioactive substances would give about  $10^{-3}$  Bk/liter of ocean water activity immediately after their disposal. This is three orders lower than the natural activity of ocean water.

Obviously enough this additional antropogenic radioactivity of the oceans does not pose any actual threat to human safety. Nevertheless the concern was expressed that even this small antropogenic radioactivity may impair the development of ocean biota because many new and unusual to Nature radioactive elements

have been dumped. Admitting the existing insufficiency of the present-day understanding of this problem it was recommended to discontinue the practise of radioactive dumps. Thus we may admit that until the magnitude of antropogenic radioactive contamination is not studied more carefully the proposed here methodology of risk accessment can not be applied. As yet we may state that probably the risk of radioactive contamination of the oceans for humans does not exceed risks of other ocean contaminants i.e. oil-products.

The situation on the local level is quite different. For instance radioactive contamination of Barents and Kara Sea is much more dangerous than the contamination of the oceans. The upper limit of antropogenic radiactivity level in this region is estimated as 10(17) Bk [2]. If these redionuclides were uniformly "stirred" in The Barents and Kara Sea aquatory the resulting water activity would be equal or even exceed its natural level. The sea currents and plancton distribution would make the situation even worse. This may be considered as a good reason for starting large-scale programme of risk assessment in this region.

## **2. Regional contamination of ocean.**

No doubt the regional radioactive contamination of ocean nowadays presents more acute problem than global contamination of the oceans (if the existing programmes of radioactive waste disposal be carried out). It is local conditions that may lead to giving the first priority to the problem of radioactive contamination of the ocean in the certain region.

Here we face some specific problems associated with risk analysis and management.

### **2.1 Absence of reliable data on contamination.**

We may point out various difficulties assotiated with obtaining data on the sources of radioactivity: dump sites, explosions, landand ship-based sources, river effluents, etc. Military secrecy also present considerable difficulties.

### **2.2 Absence of accurate methods of calculation of migration**

of radioactive substances throughout the whole "chain" from the source up to humans themselves. The absence of experimental data for verification and development of risk assessment methods results in tremendous uncertainties. This may be said of radioactivity distribution in the oceans, ocean currents, the migration patterns of oceanic biota, biological accumulation of radioactive substances.

Due to these inherent uncertainties of risk evaluation it can be applied to regional risk assessment at the present stage only to a very limited extent: it can be used at some specific stages of hazard transfer such as

a) effluent from containers, reactors, nuclear warheads into the ocean water

b) risk of radioactivity being found in sea food and products (given the level of ocean water radioactivity).

### **2.3 Differences in national priorities in risk management in various countries.**

The risk from radioactive contamination of ocean for the country-polluter may be very different from this risk for a country-"acceptor". The country-polluter may be interested in investing all its resources in risk reduction from *other* sources of hazard: this way it would minimize the risk level for its population most effectively. This situation often takes place in the problem of transboundary transfers of atmospheric pollutants. To settle down such a conflict between the two countries a special strategy should be developed.

### **2.4 How to manage and assess risk perception?**

Even if satisfactory understanding of all the problems associated with risk assessment is reached and the optimal strategy of risk management is developed the one major difficulty still remains: the risk management policy can not be successful unless the people's perception of radioactive risk and the proposed methods for its reduction is not adequate. The thing is that in radioactive risk problem various speculations and rumours

are always presented and often promulgated by commercial or<sup>429</sup> political structures. Unfortunately there are no adequate methods for assessment of socio-psychological risk associated with certain source of hazard. The existing methods of risk perception are characterized by expert and qualitative approach. And searching for optimal ways of communication between the population and the officials remains the only possible method for socio-psychological risk management.

### **3. Short-time goals and the directions of further research of the problem of radioactive contamination in Arctic.**

3.1 It is important to note here that after the preliminary identification of radioactive hazard which is now carried out (and should be continued); the next important step must be the comparison of different risks and putting forward the priorities: then we may focus our attention on the most acute risks and hazards.

Summarising the available data on radioactive contamination of Arctic [2] **we may set up the following short-time priorities:**

- Komsomolets submarine: a) total activity of the reactor zone is about 150 Kki (including 42 Kki of strontium 90 and 55 Kki of Cesium 137). b) Plutonium in the nuclear warheads: 430 Kki
- Reactor of submarine No. 412: total activity 800 Kki
- Reactors of subs No. 285, 901, 601 and of "Lenin" icebreaker: total activity 1500 Kki.

Further research should be carried out to gather more precise data on existing sources of radioactivity and to explore the oceans in order to find new and yet unknown sources of radioactive pollution. This research should be based on reviewing national archive materials and on monitoring of the aquatory.

Special attention ought to be given to difficulties associated with gathering technological data which may impair

secrecy of military objects. The only way to deal with such difficulties is to develop a methodology of gathering data for risk analysis which would allow to calculate probabilities of risk distribution and at the same time would not require disclosure of technological secrets (the content of warheads, the structure of fuel elements and submarine reactors, etc.). These data may be submitted by national institutions in the form of:

- rates of penetration of radioactive substances in ocean water;
- rates of corrosion of materials used for isolation of radioactive substances;
- isotope contents of the deposited radioactive materials.

For this purpose a special research programme should be carried out by national organizations of the interested countries. This programme includes estimation of flow rates of radioactive substances from the containers into the sea water.

**The international part of the programme should coordinate:**

- exploration of the sea currents;
- research on bioaccumulation of radioactive substances;
- monitoring of the oceans;
- estimation of corrosion rates of the materials used for isolation of radioactive materials from the environment.

3.2 Due to large uncertainties mentioned above the proposed methodology of risk assessment is applicable only at individual stages of radioactive hazard transfer:

- flow of radioactive substances from containers, reactors, warheads into the sea water;
- risk of radioactive contamination of sea food and products, given the level of water and plancton radioactivity.

3.3 Problem of transboundary transfer of hazardous substances.

The mutually acceptable variants of settling down this



problem have not been found yet. This may be said of both the routine transfers and potential transfers in case of a major accident (at NPP or in chemical industry). Many agreements regulating this problem have been signed at the UN conferences and by other international institutions, but they all proved to be ineffective in practice.

Among these problems one may mention the effluence of hazardous substances into the "common" aquatories. On our opinion, this problem should be regulated not only on the global UN level but also on regional level. The regional projects may appear to be more effective in practice and coordination.

#### 3.4 Assessment and management of socio-psychological component of risk.

Having said about various difficulties associated with this problem we may propose the following measures:

- preparing "popular" reports on radioactive contamination of the oceans, understandable for general public and local authorities.

- organizing a special project for communication and cooperation with local population. Such projects are being used in nuclear energetics in the number of countries.

3.5 The existing practice of radioactive dumps in the Arctic and Northern Atlantic presents an acute danger to the population of this region. The complexity of the problems of radioactive contamination and the need to find the optimal solutions to these problems indicate that **a special international regional project is required.**

The world community has a significant experience in carrying out international projects of this kind. A good example of such a project (with similar goals and objectives) is the cooperative programme of several international agencies known as

INTER-AGENCY PROJECT ON THE ASSESSMENT AND MANAGEMENT OF  
HEALTH AND ENVIRONMENTAL RISKS FROM ENERGY AND OTHER COMPLEX  
INDUSTRIAL SYSTEMS

UNEP, WHO, IAEA, UNIDO.

This project has been carried out already for several years and optimal methodology of researches has been proposed. Based on the existing national experience, this project has to develop the general recommendations for risk analysis and management in different regions. As the result the Guide for risk analysis has already been prepared.

We feel a strong need to work out another Inter-agency project based on the same principles. The proposed project would coordinate the research in the most heavily contaminated aquatories and may be carried out by the countries which experience the risk from the radioactive dump sites in the Arctic and Northern Atlantic region. The draft of the proposed project is attached to this report.

**References.**

1. IAEA - TECDOC-562
2. Факты и проблемы, связанные с захоронением радиоактивных отходов в морях, омывающих территорию Российской Федерации. Администрация Президента РФ. Москва, 1992г.

## INTERNATIONAL PROJECT

THE DEVELOPMENT OF SCIENTIFIC BASIS AND QUANTITATIVE DATA  
FOR RISK ANALYSIS AND MANAGEMENT TO BE TESTED IN  
THE ARCTIC AND NORTHERN ATLANTIC REGION.IAEA/UNEP/WHO/UNIDO  
RUSSIA/USA/NORWAY/CANADA/GREAT BRITAIN

## Introduction

The analysis of tendencies of the world community development shows the growth of industrial and agricultural production, power consumption, the rise of new technologies, substances and materials. As a result the state of the environment is adversely affected, human life and health danger increases. Academic community has accumulated considerable information on global environmental changes (GEC) and dangers threatening sustainable development of Nature and Humankind: greenhouse effect increase, acid rains, ozone depletion, nuclear and chemical waste disposal.

Within the activities of international organizations of UN, EEC, other international institutions and national organizations a number of researches on mentioned problems were carried out, the results of activity of different specialists and organizations all over the world are constantly generalized.

Last decades many conventions, agreements, treaties have been developed and concluded with the purpose to organize the system of technological activity risk management at different levels: global, transboundary, national. At the same time analysis of these agreements, conventions and treaties shows their low efficiency. For example the report of the intergovernmental group of WHO and UNEP experts on climate changes

says: "Consideration of strategies to respond climate changes is very difficult for politicians. Available information is insufficient for deep political analysis due to:

- uncertainty concerning efficiency of specific variant or group of variants of response for actual counteracting to potential climate changes;

- uncertainty concerning costs, economic growth and other economic and social consequences of specific variants or group of variants of response".

The principal objective difficulties in such agreements and treaties development and realization is different economic, social and political situation of the countries as well as other characteristics related to geographical position, national peculiarities, historical development, etc.

One has succeeded in developing a basis to establish a parity in interstate relations on similar problems in the field of military, political and trade relations on the basis of the last Humankind experience. However, at present acute need to establish the same parity in the field of ecology is felt.

The global "environmental parity" concept includes the development of scientific basis for risk management and international norms to regulate disposal and accumulation of radioactive substances in oceans (RASO). The proposed here RASO project initiates subsequent bilateral and multilateral agreements between the interested countries. As a result not only radioactive contamination will be cleaned up but also the global risk of radioactive pollution will be reduced. Thus we may avoid many unpredictable consequences and therefore contribute to already existing international programmes.

### ***The main objectives of the project***

Organize monitoring of dumping and transfer of RASO from the existing sources.

Develop and co-ordinate criteria of settling accounts for damage in transboundary transfers of RASO.

Develop methodology of damage assessment in transboundary transfers of RASO.

Assess possible mutual settlements and to work out proposals<sup>435</sup> on different variants of settling and payment.

Work out in details and to assess acceptability of propositions of existing projects on radioactive waste disposal.

Develop scientific-methodological recommendations for possible agreements on transboundary transfers of RASO in the Arctic and Northern Atlantic regions.

### ***The main parts and stages of the project. \****

#### General and regional problems:

- Principal co-ordination with all the interested parties of the RASO project.

- Revise the RASO project and sign it by all the interested parties (final text will be corrected as the stages of the project are being carried out).

- Develop a detailed schedule for carrying out the consequent stages of the project.

- Locate in the Arctic and Northern Atlantic regions certain areas which correspond to the purposes of the existing Inter-Agency Project on Risk Management in Large Industrial Areas.

- Carry out researches in these areas in accordance with mutually agreed procedure and methodologies.

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\* The terms of the fulfilment of stages should be determined after making detailed plan of works on the project.

#### Transboundary problems and radioactivity sources.

- Carry out researches on identification and mutual transfer of hazardous substances across the border and methodology of these researches.

- Carry out researches on the size of damages arising from transboundary transfers using developed and co-ordinated methodologies.

- Review the existing methodologies for identification of RASO and develop transfer models.

- Develop concept, criteria and methodology of damage

compensation as a result of transboundary transfers.

- Carry out assessments on possible mutual settlements for transboundary transfers of hazardous substances in routine and potential transfers.

- Work out recommendations on possible agreements on transboundary transfers.

- To work out a guide on transboundary transfer risk management.

The above mentioned guide and recommendations should be considered as preliminary for this project. They will require further research taken for various groups of countries.

### ***Organizational structure of the project***

The Project is realized as a joint project of international organizations IAEA, UNIDO, WHO, UNEP and organizations of Russia, Norway, Great Britain, USA and Canada. To study applied problems of transboundary transfers certain areas are to be chosen.

The chosen areas in the Arctic and Northern Atlantic regions according to the request of their governments are included as separate independent regions into the existing Inter-Agency Project on Risk Management in Large Industrial Areas of IAEA, WHO, UNEP, UNIDO. Research and organizational work in these bordering regions is coordinated by a specially appointed organizations and institutions. On the basis of data obtained in stage-by-stage realization of the project these organizations and the participating countries compile results elicited at all stages of the project.

The following organizations and institutes carry out the project:

From Russia:

- Ministry of Ecology and Natural Resources, Ministry of Science, State Committee on Civil Defence, Emergency Situations and Liquidation of Consequences of Disasters, State on Radiation and Nuclear Safety under the President of Russia, Academy of Sciences of Russia and their organizations in the studied regions under the support of Committee on Ecology and Rational Utilization of Natural Resources of the Supreme Soviet of Russia;

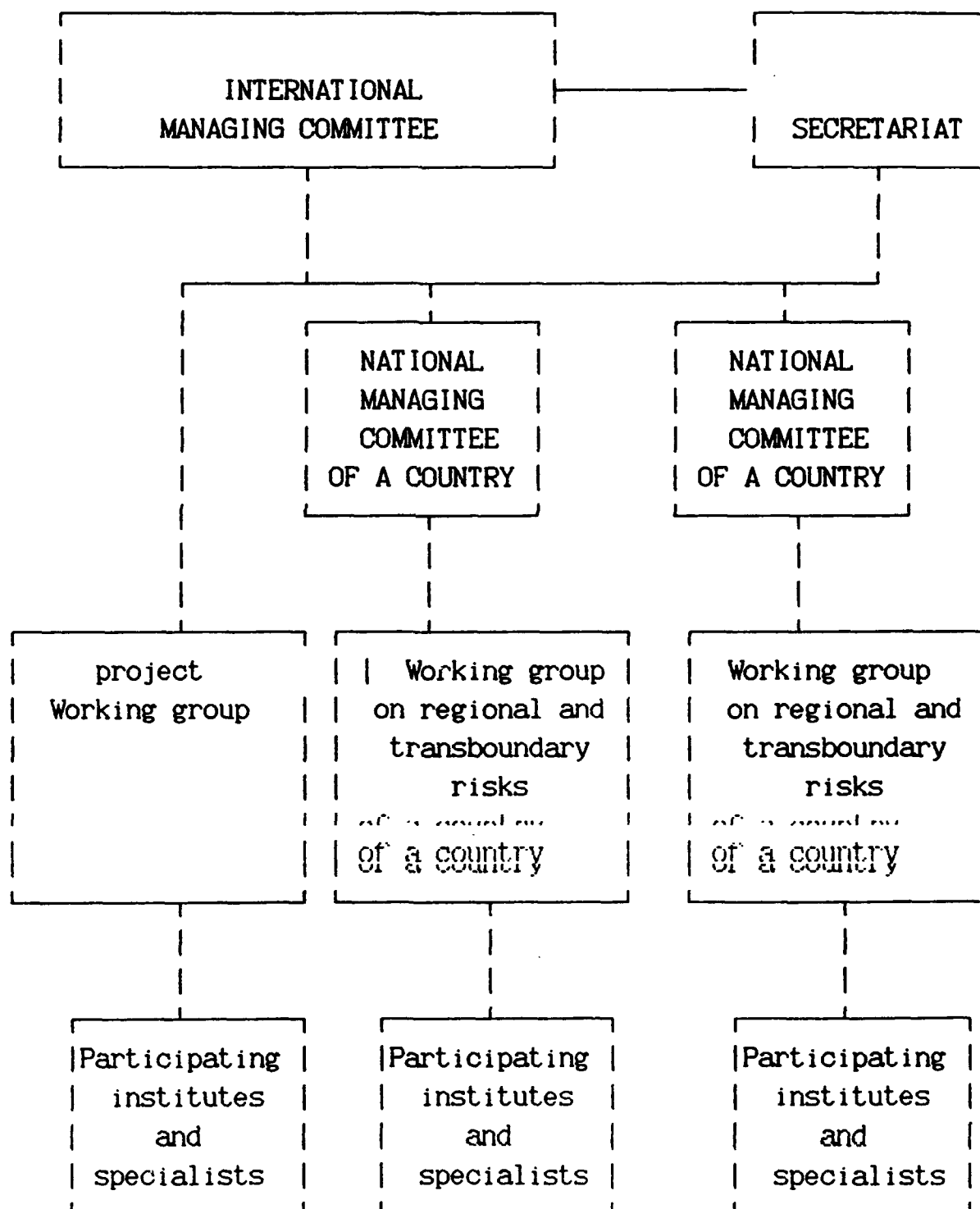
- Local authorities in the regions;

- Specialists from scientific-research institutes and<sup>437</sup> centers.

From other countries: will be defined later.

**Management of the project**

Management of the project is realized according to the following framework:



*Financial support for the project*

Organizations and ministers participating in the project provide financial support for their specialists working on the project. They also provide financial support to co-ordinate and carry out works in experimental regions. The leadership of the project will aspire to provide additional financing at the expense of contributions of other national and international foundations and organizations which are not the project executors.



## THE EXPEDITION TO THE NUCLEAR SUBMARINE *KOMSOMOLETS* SITE

by

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As it is known, the nuclear submarine *Komsomolets* of the Soviet Navy had been lost on April 7, 1989, in the Norwegian Sea, 300 miles off the Norwegian coast.

The principal particulars of the submarine are as follows:

weight:	about 6000 tons
length:	about 120 m
hull diameter:	11 m
hull material:	titanium alloys

The submarine has onboard one water-cooled/water-moderated reactor and two torpedoes with nuclear war-heads along with conventional torpedoes.

The disaster that happened to us had been a horrifying, painful shock to all honest people in the world. Ardent discussions of this tragedy have not ceased till now.

The public is concerned about the ever-deteriorating state of global environment and, quite naturally, another "hot spot" in the ocean is stirring up many minds and urges the governments to take efficient actions.

My country's government tried to analyze the existing situation as far back as immediately after the Chernobyl tragedy, but the "Komsomolets" disaster has only aggravated the problem.

In order to clarify the true causes of what happened aboard the *Komsomolets*, and with regard to all collected data and the attitudes in the society, the Government Act of the recovery of the submarine *Komsomolets* was issued in 1990.

The mission of the recovery of the submarine was assigned to the Central Design Bureau for Marine Engineering (CDB ME) "Rubin" which had designed the submarine, and is the head organization in the development of the deep-water vehicles in our country.

Various organizations in my country and foreign companies submitted several dozens of proposals as to the procedure of recovery. The most realistic project which could be implemented within the required period of time (1991 to 1992) was the project of several Dutch firms which established the Netherlands Deep-Water Operations Consortium (NDOC) for that purpose. The management of all works for the Russian side is carried out by the Academician I.D. Spassky. The preparatory activities in 1991-1992 involved a large-scale complex of research and other work fulfilled by home and foreign enterprise, institutions and companies.

To make up for the lacking data on the submarine's state, to specify the radiation situation and to study the motions of water masses in the sea area at the site, a number of expeditions as a preparatory activity for the recovery was held from April to September of 1991, involving seven ships of different departments. The core of this programme was the expedition of the research vessel *Akademik Mstislav Keldysh* with two deep-water manned vehicles "Mir" aboard. One more expedition was held in 1992 which consisted of two ships: the research vessel *Akademik Mstislav Keldysh* and oceanographic research vessel *Ivan Kruzenshtern*.

Without dwelling at length on this action, I would only note that the CDB ME specialists participated in dives of the "Mir" vehicles to the submarine along with the regular pilots, which allowed to obtain very important data.

1. Total Scope of Works Done by the Expedition:

- number of divers of the "Mir" vehicles—12;
- total submerged time of the "Mir" vehicles—120 hours;
- number of measurement recordings—about 2000;
- number of bottom soil and water samples taken by the "Mir" vehicles—200.

2. Results of the Expedition

The expedition had accumulated a large amount of facts. During the examination it was determined that the radiation situation at the submarine location was normal.

The models of possible development of events worked out by the scientists were proved in principal and forecasts updated.

The nuclear reactor itself is safely shutdown, however, it does contain sea water and it, quite naturally, is radioactive. The rate of the radionuclide escape from the reactor, as confirmed by on-site measurements, is low and, therefore, a small leakage of radioactive substances at the present time is not dangerous for people and environment.

Torpedoes with nuclear war-heads are also exposed to sea water effects due to corrosion processes, however, no noticeable leakage of radionuclides therefrom was recorded.

It should be noted that, approximately from 1995, escape of plutonium to the marine environment due to corrosion of the reactor vessel and torpedo body metal is likely to start which may affect the environmental situation in the area. It is this prediction that necessitates systematic monitoring of dynamics of the processes inside the submarine and in her vicinity, which requires considerable expenditures to arrange and outfit expeditions. The results of the examination of the pressure hull, which was found to

have significant damages and fractures in the forward portion of the submarine, were rather important and somewhat unexpected. This circumstance has demanded considerable alterations of plans of works on the submarine *Komsomolets* and additional preparatory works the most important of which are experiments on patching (sealing), in 1991-1994, of some openings through which radioactive substances may escape into sea water. Very intensive works are carried out to study the methods of extraction of the nuclear ammunition from the submarine, may be with torpedo tubes as one piece.

And one more thing. A great number of articles like clocks, spoons, bolts, hydrophones, etc., were recovered from the submarine and from the sea bottom in her vicinity and are now exhibited at the Central Naval Museum, St.Petersburg. All these articles were checked for radioactivity, no deviations from the background level or normatives were detected.

Third expedition to the submarine *Komsomolets* is planned to be carried out in July, 1993. The expedition's aim is to study once again a number of issues which allow to make the final decision pertaining to the fate of the nuclear ammunition of the submarine.

Thank you for your kind attention.

Radioactivity and Environmental  
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**International Arctic Seas Assessment Project  
IASAP (1993-1996)**

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**1. INTRODUCTION**

According to official information from the Russian Federation [1, 2], high, medium and low level radioactive wastes were dumped into the Arctic Seas during the years 1959-1992. The total amount of radioactivity dumped is more than 90 PBq, including seven nuclear submarine or icebreaker reactors with fuel containing a total of 85 PBq; ten reactors without fuel containing 3.7 PBq; liquid low level waste containing 0.9 PBq, and solid medium and low level waste containing 0.6 PBq.

The packaged solid wastes and nuclear reactors were dumped in the Kara Sea in the shallow bays of Novaya Zemlya, where the depths of the dumping sites range from 12 to 135 m, and in the trough of Novaya Zemlya at a depth of 380 m. Most of the low level liquid wastes were discharged in the open Barents Sea.

Concern has been expressed in many quarters over the possible health and environmental effects of the dumped radioactive wastes. To address this concern and to co-ordinate international efforts, the IAEA has established the International Arctic Seas Assessment Project, IASAP. The project was launched in February 1993 in Oslo, at a meeting which was attended by 60 experts from 12 countries and 4 international organizations. It was organized by the IAEA in co-operation with the Norwegian and Russian Governments.

The Norwegian/Russian expert group plans to arrange exploratory cruises to the actual dumping sites in June-August 1993 and in later years, if necessary. The IAEA Marine Environment Laboratory (MEL) will participate in the cruises and will be involved in the analyses of environmental samples.

**2. OBJECTIVES OF THE PROJECT**

The objectives of the project are:

1. To assess the risks to human health and to the environment associated with the radioactive waste dumped in the Kara and Barents Seas.
2. To examine possible remedial actions related to the dumped wastes and to advise

on whether they are necessary and justified.

The project was endorsed by Contracting Parties to the London Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter at the 15th Consultative Meeting in November 1992. The IAEA was requested to report its findings to the Convention at the earliest opportunity.

### 3. PROJECT IMPLEMENTATION

The project will be conducted in the setting of the four working groups established at the Oslo meeting:

- 1) Source terms,
- 2) Existing environmental concentrations,
- 3) Transfer mechanisms and models,
- 4) Impact assessment and remedial measures.

In the beginning, emphasis will be put on the first, second and third groups. The importance of the fourth group will increase, as information from the other groups becomes available.

The work will be carried out by using the normal IAEA mechanisms:

- Consultants, Advisory Group and Technical Committee meetings,
- a Co-ordinated Research Programme,
- Research and Technical contracts.

### 4. WORK PLAN

#### 4.1 Source term

Basic information on the source term was given in the reports of the Russian Federation [1, 2], but detailed information on the radionuclide composition of the dumped waste or on the characteristics of the fuel in the different types of reactors dumped was not included. Another group of questions concerns the protection barriers provided for the dumped reactors. Most of them are shielded with metal or concrete and filled with furfural, a kind of polymer. It has been stated that they are safe for several hundreds of years. Information on all of these matters has to be investigated using archive sources or will have to be reconstructed in order to provide data to enable source term calculations to be made. For this purpose, contracts with Russian institutes will be concluded.

#### Tasks for 1993

- Study of nuclide composition and characteristics of fuel in dumped reactors.
- Study and description of the state of protection barriers of the dumped reactors and other packaged waste at the time of dumping and after extended immersion.
- Visual investigation of reactors and other wastes during the 1993 cruise and later if necessary.
- Sediment sampling and in situ measurements during the 1993 cruise and later if necessary. Sediment analyses will be carried out in several laboratories.
- Review of progress on source term evaluation.

#### Tasks for 1994

- Theoretical assessment of release rates of various radionuclides from different types of dumped reactors and shielding.
- Description of the typical composition of dumped low and intermediate level waste.
- Collection of information on the physical and chemical characteristics of furfural.
- Investigation of the stability of furfural against radiation, heat, saline water etc. (experimental studies).
- Review of progress of source term evaluation.

#### Task for 1995

- Finalization of source term evaluation and production of a report.

#### **4.2 Existing environmental concentrations**

Information on the levels of radioactive contamination in the target area and other areas of the Arctic seas will be collected as input to a global data base which is under development at IAEA-MEL. A questionnaire has been sent to all relevant laboratories requesting the appropriate data. The reliability of the data will require evaluation.

#### Tasks for 1993

- Development the data base and acquisition the existing environmental data from the various laboratories.

#### Task for 1994

- Review of completeness and assessment of the reliability of the environmental data.

#### **4.3 Transfer mechanisms and models**

An IAEA Co-ordinated Research Programme (CRP) entitled "Modelling of the radiological impact of radioactive waste dumping in the Arctic Seas" has been established. Several national laboratories and IAEA-MEL will participate in the CRP. The objective of the CRP is to develop realistic and reliable assessment models for the Arctic Sea areas and to co-ordinate the efforts of different laboratories in the field.

A staged approach to the final modelling assessment will be taken. In the early stages, when data is incomplete, a relatively simple preliminary modelling exercise will be conducted using artificial source terms. As more information becomes available the exercises will become increasingly realistic. A group of key modellers has already met to plan the CRP.

A great deal of information is needed on the oceanographical and ecological features of the area, as well as on the living habits of possible critical groups. This information will be obtained through contracts with Russian institutes.

Tasks for 1993

- Planning the preliminary phase of the international modelling exercise (CRP).
- Preliminary modelling using unit release, a simplified oceanographic environment and standardized transfer data.
- Determination of site specific environmental data and living habits.
- Obtaining site specific oceanographic data.

Tasks for 1994

- 1st RCM of Modelling CRP
  - comparison and documentation of results of first modelling exercise (unit release),
  - creation of second modelling exercise with available source term and environmental information,
- Continued acquisition and improvement of source term and environmental information.

Tasks for 1995

- 2nd RCM of Modelling CRP.
- Comparison assessment and documentation of model predictions and creation of final modelling exercise.

Tasks for 1996

- 3rd and final RCM of Modelling CRP.
- Compilation and analysis of final results and preparation of report.

**4.4 Impact assessment and remedial measures**

An impact assessment working group will lead and advise the other groups. It will follow and monitor the information stream from the source term group, the results of the site specific ecological studies, and it will review the progress of the modelling group. It will also evaluate possible remedial actions, their need and feasibility.

Tasks for 1994

- Review of possible engineering solutions for remediation,
- Review of results of the investigations on source term and preliminary assessments made by modelling group.

Task for 1995

- Review of modelling assessments and further consideration of remedial actions.

Task for 1996

- Finalization of the assessment report and preparation of recommendations.



#### 4.5 Project review meetings

- 1994 - Review the working plan and progress.
- 1995 - Review of progress and group interaction to finalize the work.
- 1996 - Finalization of work and review of report.

#### 4.6 Final report

Report completed in 1996 for presentation to London Convention 1972.

### 5. CONCLUSION

The dumping of radioactive waste in the Arctic Seas has caused concern in many countries and has stimulated research and assessment work in national laboratories. The IAEA has established an international project in this topic area.

The project aims to produce an independent and objective assessment of the potential radiological implications of the dumping and to address the question of possible remedial measures. At the same time, it is intended to provide a focus for reporting national research and assessment work and a mechanism for encouraging international co-operation and collaboration.

### 6. REFERENCES

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## ASSESSMENT OF DEEP-SEA NEKTON AND THEIR ROLE IN RADIONUCLIDE TRANSFER

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Since the development of nuclear power and weapons, radioactive waste disposal has become a serious problem. There have been many documented instances of deliberate and accidental release of radionuclides in the sea. In circumstances where these wastes are uncontained by design, by accident, or by passage of time, the radionuclides released are known to contaminate the sea bottom, the waters that flow over it, and the organisms living there.

Under present waste disposal practices, (i.e. dumping drums or loose waste, release via pipeline, or accidental release via ship sinkings) there is guaranteed release; therefore contamination is a problem to be considered, assessed, and reduced or contained if possible. The realities are that reduction or containment of wastes already dumped in the sea is difficult, dangerous, and expensive. Therefore, it seems most likely that assessment of the dangers to the environment, and reduction of potential exposures of humans to released radionuclides are the most positive steps that can be taken regarding the already released materials.

Sampling deep-sea animals has always been expensive, time-consuming, and difficult (Mills, 1983; Pearcy, 1975; Rowe and Sibuet, 1983). Many deep water organisms have been almost impossible to collect, either because they easily avoid the small nets usually used, or because they are easily damaged and cannot be recovered intact. These problems can be minimized through use of a variety of techniques and equipment, but they are always a consideration.

The marine animal community consists of three groups: benthos, nekton, and plankton. Benthos are those animals that live on the bottom or closely associated with it; plankton are, relatively speaking, drifters that are at the mercy of the currents; and nekton are strong swimmers. Nekton includes fishes, shrimps, squids and octopods, and marine mammals. These animals frequently differ widely in almost all aspects of their size, biology, ecology, and behavior. This review will concentrate on the nekton, although much of its content applies to the other groups, which are discussed briefly. Within the lifetimes of individuals of many

species, individuals may belong to more than one of the groups - possibly all of them. Each group generally has a different susceptibility to contamination, but the likelihood of individual species being contaminated depends upon their life history and the physical conditions at the contaminated site.

Benthos are most likely to be affected by waste because they are on the bottom and often live in or feed on the sediments. In deep water, they are relatively unlikely to be directly exploited by man, but because many are mobile and because they are part of the food chain for larger, more mobile predators, they function as radionuclide vectors.

Plankton are least likely to be affected because they drift with the currents and are thus (owing to dilution effects) less likely to be in contact with the more highly concentrated wastes and are also unlikely to remain in the contaminated area. It seems very unlikely that they will be directly exploited by man in the foreseeable future. Finally, their abundance in very deep water is probably relatively low.

Nekton are likely to be affected because they may remain in the area, especially if the waste containers provide substrate for food, in which case nekton may also act as concentrators (Stein, 1980). They are most likely to be exploited by man. Presently, fishes of upper and mid-slope depths such as grenadier (Family Macrouridae), orange roughy (Family Hoplostethidae), and sablefish (Family Anoplopomatidae) are important commercial species used for human consumption. The contamination of such species at an actual site, the Farallon Islands near San Francisco, California, is being discussed at this meeting.

Each of these ecologically-defined groups presents different opportunities and problems in collection. Because mobile benthic animals are usually unable to move rapidly, they are more easily collected by the small nets normally used at bathyal and greater depths. Sessile organisms may be collected by grab or corer. However, there may be some avoidance from animals capable of swimming (some holothurians - sea cucumbers) or pushed out of the way by the net itself. For some kinds of surveys such as habitat use or micro-distribution, for precise sampling, or for sampling in locations where nets cannot be used, undersea vehicles such as ROV's and submersibles are required. Plankton are easily captured using relatively small nets, but it is frequently difficult to accumulate enough material due to low population densities at great depths.

The nekton are hardest to collect. They are highly mobile and range in size from less than 10 cm to over 10 m. Some of the largest deep water animals, the giant squids and sharks, may in fact be almost uncollectable with present technology, although they can be photographed if they can be found. On the other hand it is

possible that, because of the relatively low populations of the largest species, they can be assumed to play an insignificant role as radioisotope vectors. Nekton are often wide-ranging and may move into and out of the contaminated zone, resulting in our inability to know what the exposure of the animals collected near the contaminants has been. Again, because of the mobility of nekton, exposure and possibility of contamination is highly variable and will depend upon factors such as diet, migratory patterns, food search strategies, schooling behavior, and other life history patterns (Stein, 1980). Finally, nekton may transport radioisotopes in all three dimensions and act as highly effective vectors. This is particularly true of species that make regular diurnal, seasonal, or reproductive migrations.

Nekton are difficult to capture with nets because of avoidance and escapement. They are difficult to detect and to see. Thus, particularly in the deep sea, it is hard to obtain truly representative samples. To sample nekton effectively requires a variety of strategies. There is no single "best method". Rather, for a given species, there is suite of methods that depend not only upon the size, behavior, and natural history of the species, but upon the life history stage of interest to the investigator and the information being gathered about it.

#### Nets large and small

Large nets collect many animals that are good at avoiding smaller nets. Compared to the smaller nets customarily used in the deep sea (Figure 1), they filter huge amounts of water ( $10^6$  m<sup>3</sup> in a single tow is not unusual); consequently they are time-efficient and statistically useful (more likely to capture relatively rare animals), but they require specialized equipment (winches, tapered wire, etc) not readily available on most oceanographic research vessels (Stein, 1985). In many ways they can be instrumented easily because their size provides space for instrument packages and stability from the effects of the size and or weight of those instruments. At a minimum, accurate time-depth recorders are needed to show where the net fished, when, and for how long. These data can be used to determine capture depth of the catch and volume of water filtered, thus providing the basis for quantitative population estimates. They can also be fitted with acoustic transponders, side-scan sonar, cameras and other equipment, much of which is easily modified from commercially available fishing gear.

There is a wide variety of large nets available for commercial purposes. These are generally unsuitable for research because their mesh is too large to collect the smaller nekton and they produce too much drag to tow if the mesh size is reduced. Smaller pelagic trawls with fine mesh (i.e. less than 25 mm stretch) have been built and used successfully to depths below 3000 meters (Stein, 1985). Such nets of 100 to 200 m<sup>2</sup> mouth area can also be

equipped with opening-closing cod ends as necessary in order to obtain depth stratified samples. Similarly, it is possible to tow large small-mesh bottom trawls at great depths. However, as with all nets towed with such a great length of warp (scope below 3000 m depth is 1.5:1 or less), it is extremely difficult to make hauls that are precisely located in relation to objects on the bottom (e.g. waste canisters). Thus, there is a substantial risk of collecting radioactive waste in nets used to sample close to dump sites.

### Undersea vehicles

Undersea vehicles (submersibles, Remotely Operated Vehicles, Autonomous Undersea Vehicles) will be very useful in waste disposal studies. They are capable of collecting real-time visual samples and visual and material samples for later analysis. All undersea vehicles suffer from similar problems (Stein, 1991). Like nets, they are often avoided by animals that can do so; sometimes they also attract animals (Stein, personal observations). The lights, thruster noise, pressure wave in front of the vehicle and other sensory cues definitely affect behavior of certain species. There is still no camera as effective as the human eye, although present-day video cameras can be extremely good, especially if the images are computer enhanced. However, no camera of which I am aware can effectively track a suddenly appearing, fast moving organism swimming laterally across the field of view.

Present deep submersibles are not very useful for obtaining more than presence-absence data about nekton, although they can effectively survey and sample benthos. There are several reasons for this problem. First, the observers typically do not have the best view - that is reserved, justifiably, for the pilot. Thus, the observers see off to the side and are more likely to miss seeing nektonic animals before they avoid the vehicle. Second, they view a lateral path whose width is difficult to measure and in which the size of organisms is difficult to determine. These characteristics make quantitative measurements harder to obtain. In manned submersibles where the observer looks directly forward and down, it is possible to obtain real quantitative estimates of fish and invertebrate populations, to map substrate types, and to correlate the distributions of fishes and invertebrates with substrate type (Table 1; Stein et al., 1992). Additionally, it is difficult to collect nekton with submersibles because of avoidance. A final significant problem in using manned vehicles is their limited bottom time, weather dependency, and expense of operation (Rowe and Sibuet, 1983).

Deep ROV's will be more useful for discrete quantitative surveys in deep water than manned submersibles (Stein, 1991). They can be equipped with a variety of cameras (low light, black and white, color, video or still or combinations of both), computer programmed

to maintain a constant altitude above the bottom and a constant speed over the ground, and some can even tow or mount small nets. In addition they may have several manipulators of different functions. Because they are unmanned and tethered, they have very long potential bottom time - days rather than the few hours typical of manned vehicles. Furthermore, because they do not use batteries but rather are surface-powered, they can mount and use more equipment simultaneously than manned vehicles without compromising their available working time. With adequate video and still cameras, it is possible to obtain quantitative data similar to that described above (Malatesta et al., 1992). They suffer the same disadvantages regarding collection of nekton as manned submersibles, but because dive duration is not as critical they can be used for collections that are very time-consuming such as deploying baited traps or hooks and waiting for a catch.

#### Traps and cameras

Traps and cameras are passive, as opposed to active, means of data collection. They can be deployed as free vehicles to be retrieved by using an acoustic anchor release, and various functions can be controlled with an acoustic transponder (Rowe and Sibuet, 1983). Depending upon the type of information to be collected, they can either be baited or unbaited. Either way, they can be very effective means of sampling nekton. In their use, many variables need to be taken into account. These include, but are not limited to: bait type, trap type and mesh size, camera field of view, and need for stereo views to determine sizes of organisms photographed. Cameras can collect a variety of data, especially in conjunction with current meters (behavior, relative abundances, and possibly absolute abundances) but cannot collect animals for positive identification. Therefore, cameras and traps used in conjunction with one another are more useful if circumstances allow.

#### Longlines

Benthic longlines, either horizontal or vertical, can also be effective collection devices, particularly for very large nekton. They can be retrieved similarly to other free vehicles, although specialized equipment may be needed to haul them if really large animals are caught. However, they, like traps and baited cameras, are highly selective. In addition to the problems involved in bait selection they have the added problem of sample (organism) loss before or during retrieval. For some organisms, however, such as large sharks and fishes, there may be no other practical method of collection.

### Why it is important to sample

Nekton may act as an important radionuclide transferral mechanism to humans (Stein, 1980). Most, if not all, of the world's easily exploitable fish populations are at their limit of sustainable use or beyond it. As the world's more easily exploited stocks are over-fished, deeper usable species are fished (orange roughy, blue grenadier, Pacific grenadier, roughscale grenadier, thornyheads). Fishing for these species in regions where there has been radioactive waste disposal increases the likelihood of transferral to humans.

Nekton may spread radionuclides far more widely and less predictably than benthos or ocean currents. Many nektonic species make extensive migrations for feeding or spawning. These can be either geographic or vertical. In some cases, deep-water bottom fishes may occur far offshore in the pelagic within their depth zone but well above the bottom (Stein, 1985). Many nektonic species have differential distribution by sex, size, or age. It is possible, although unlikely, that contaminated animals that have broadcast spawning with pelagic eggs and juveniles provide a mechanism for direct, rapid transport of contaminants from the spawning depths to near the surface (Stein, 1980). Thus, knowledge of the natural history and behavior of the animals in the area is often necessary to be able to trace and predict occurrence and transport paths for contaminants.

Our knowledge of the deep pelagic and deep-sea in general is not very good; some evidence exists that there is more active transport between the bottom and midwaters than thought (Stein, 1985). We need to know (especially given the known wastes already disposed of in the sea) if the "dilution solution" is effective. Does the radioactivity have any effects on organisms, either near or far field? The one known effect of which I am aware is that the creation of hard substrate in areas of soft bottom where it did not previously exist provides surfaces for sessile animals that may serve as food, and thus as attractants for, nektonic predators. The dumped material itself becomes an artificial reef populated by organisms very likely to become contaminated.

Some evidence exists that the traditional view of the deep-sea pelagic zone as a desert may be incorrect, and that more transport between the bottom and midwaters exists than previously thought. Off Oregon, USA, a nominally benthic species of macrourid, Coryphaenoides filifer, usually collected between 2200 and 3100 meters in bottom trawls, comprised 55% of the total fish biomass collected in midwater below 2000 meters (Stein, 1985) - above a bottom 4200 meters deep (Figure 2). Apparently the species can live in midwater far offshore although it remains within its "normal" depth range. It has been suggested, but not established, that there are diurnal migrations at abyssal depths, in which animals migrate up into the water column. Clearly, if such



migrations exist, they will affect the speed and distribution of radioactive contamination.

How can data on deep nekton be best obtained?

There is no single best sampling method. All of the gear discussed above is selective. Therefore, the best way to acquire the most accurate view possible is to use a combination of methods. By designing a sampling plan that will collect from as many of the groups as possible, a composite picture can be assembled. It is very important to remember that one has no idea of the animals one does not see, and it is consequently unwise to assume that a complete sample exists.

Nets (especially large ones) are particularly effective in midwater, where organism densities are relatively low and thus the animals are relatively unlikely to be seen from vehicles. Furthermore, nets provide material for positive identification and for tissue analysis. Manned vehicles are useful for manipulative experiments and qualitative surveys, but in deep vehicles observer ability to see is not usually very good. However, using a vehicle equipped with good side-scan sonar capabilities might allow detecting, tracking, and viewing animals. Unmanned vehicles should be able to perform excellent surveys when equipped with appropriate cameras, etc. They can have up to 3 real-time video cameras, constant altitude, color side-scan sonar and other equipment. The side-scan may be useful for estimating avoidance. The view forward should also be good straight ahead. Collection of the largest deep nekton (i.e. giant squid for instance) is problematic. Their spatial distribution is unknown, their occurrence is unpredictable and they are very large (hundreds of kilograms). Perhaps the best answer to that problem is simply to assume their size and position at the top of the food chain means they are rare and thus unlikely to have a significant role in contaminant distribution.

It is unwise to base sampling plans on assumed behavior of nekton. Behavior of animals is not necessarily intuitively obvious. For instance, the orientations of shrimp and fishes in relation to water current direction at a baited camera at 5000 meters on Hatteras Abyssal Plain were statistically different; the fishes were much more strongly oriented into the current than were the shrimps (Figure 3; Stein and Rowe, unpub.). Behavior related to current may be affected or complicated by biological interactions (Lampitt et al., 1983). There are usable models that allow prediction or estimation of populations based upon odor diffusion and physical oceanographic principles (Rowe et al., 1986). Such models, still developmental, could allow prediction of populations in a way fundamentally different from other methods because they do not require physical sampling of the organisms and are relatively free from the influences present with powered vehicles.

Thus, one might design a sampling program to study the fauna at a known dump site by using large opening-closing trawls above the site, large bottom trawls more than 0.5 km from the waste, unmanned vehicles to make detailed surveys of the organisms within that distance, traps or long lines near or actually within the waste field, and or unmanned vehicles to sample the wastes themselves and the closely associated organisms. Adequate planning is most important; it is a necessity to know exactly what the study goals are, because that will determine sample sizes needed, life stages to be sampled, location of sampling, and other factors. On these bases, the type of gear can be selected that is most likely to provide the desired data.

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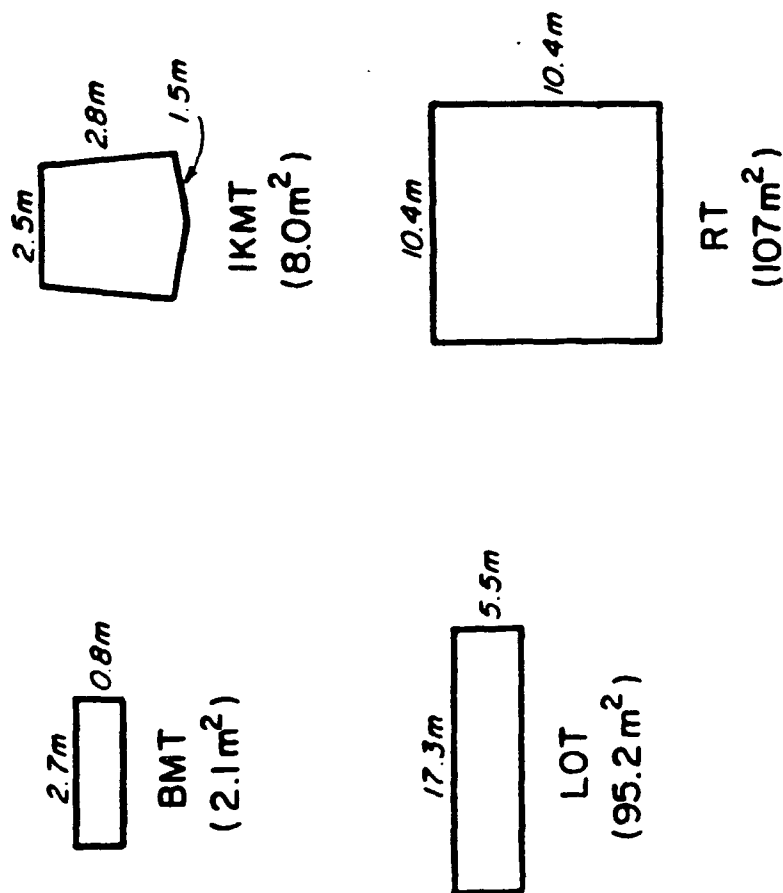


Fig. 1. Fishing dimensions and effective mouth areas of 3-m Beam Trawl (BMT), 3-m Isaacs-Kidd Midwater Trawl (IKMT), North Sea Otter Trawl (LOT), and pelagic Rope Trawl (RT).

Table 1

Average number of fish per hectare ( $10^4 \text{ m}^2$ ) on the seven most distinct habitat types, as determined by cluster analysis (see Fig. 4). Only the 21 most abundant taxa are listed, these taxa used in the canonical correlation analysis. Most-abundant taxon in each category underlined in bold characters. Species absent from a specific habitat are indicated with dashes.

Species	Mud	Mud & cobble	Mud & boulder	Cobble	Boulder	Flat rock	Rock ridge
Agonidae	186	464	1122	—	25	—	18
Bathymasteridae	7	7	—	—	—	—	15
Big skate	7	—	51	—	—	—	—
Canary rockfish	—	14	102	—	—	158	82
Cottidae	24	79	51	67	—	158	73
Dover sole	<b>499</b>	343	2295	—	—	—	15
Greenstriped rockfish	64	364	204	266	25	—	79
Kelp greenling	—	—	—	67	76	316	27
Lingcod	—	—	—	67	—	—	30
Longnose skate	7	14	51	—	—	—	6
Pygmy rockfish	21	2129	<b>8926</b>	<b>999</b>	<b>2772</b>	—	<b>1785</b>
Redstripe rockfish	—	7	—	—	—	—	43
Rex sole	107	57	1887	—	—	—	—
Rosethorn rockfish	26	343	408	933	161	<b>474</b>	675
Sharpchin rockfish	60	<b>2930</b>	2754	133	—	—	277
Shortspine thornyhead	239	<b>443</b>	2193	—	—	—	—
Slender sole	76	107	408	—	—	—	—
Spotted ratfish	26	14	510	—	—	—	—
Yelloweye rockfish	—	7	—	—	25	—	27
Yellowtail rockfish	—	29	—	67	176	—	191
Zoaridae	282	279	1887	—	50	—	18

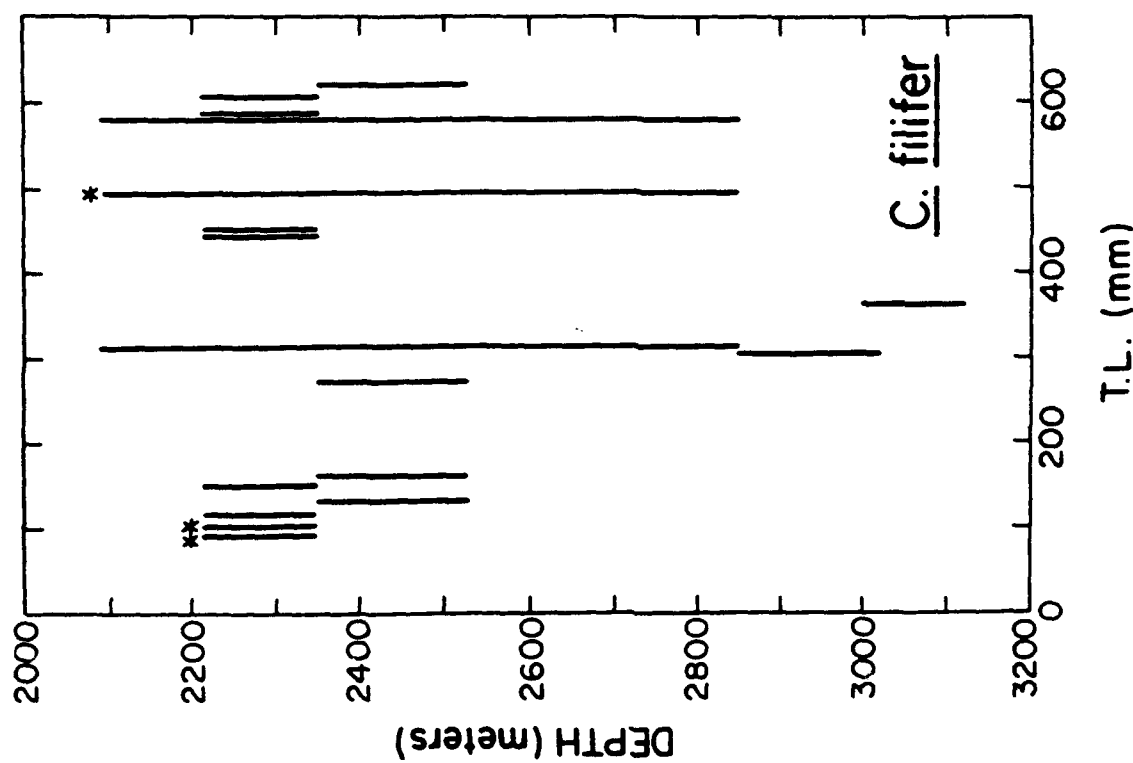


Fig. 2. Depth of capture vs total length (TL) for individual *Coryphaenoides filifer* collected in nets at depth. Asterisks indicate individuals missing the terminal portion of the tail.

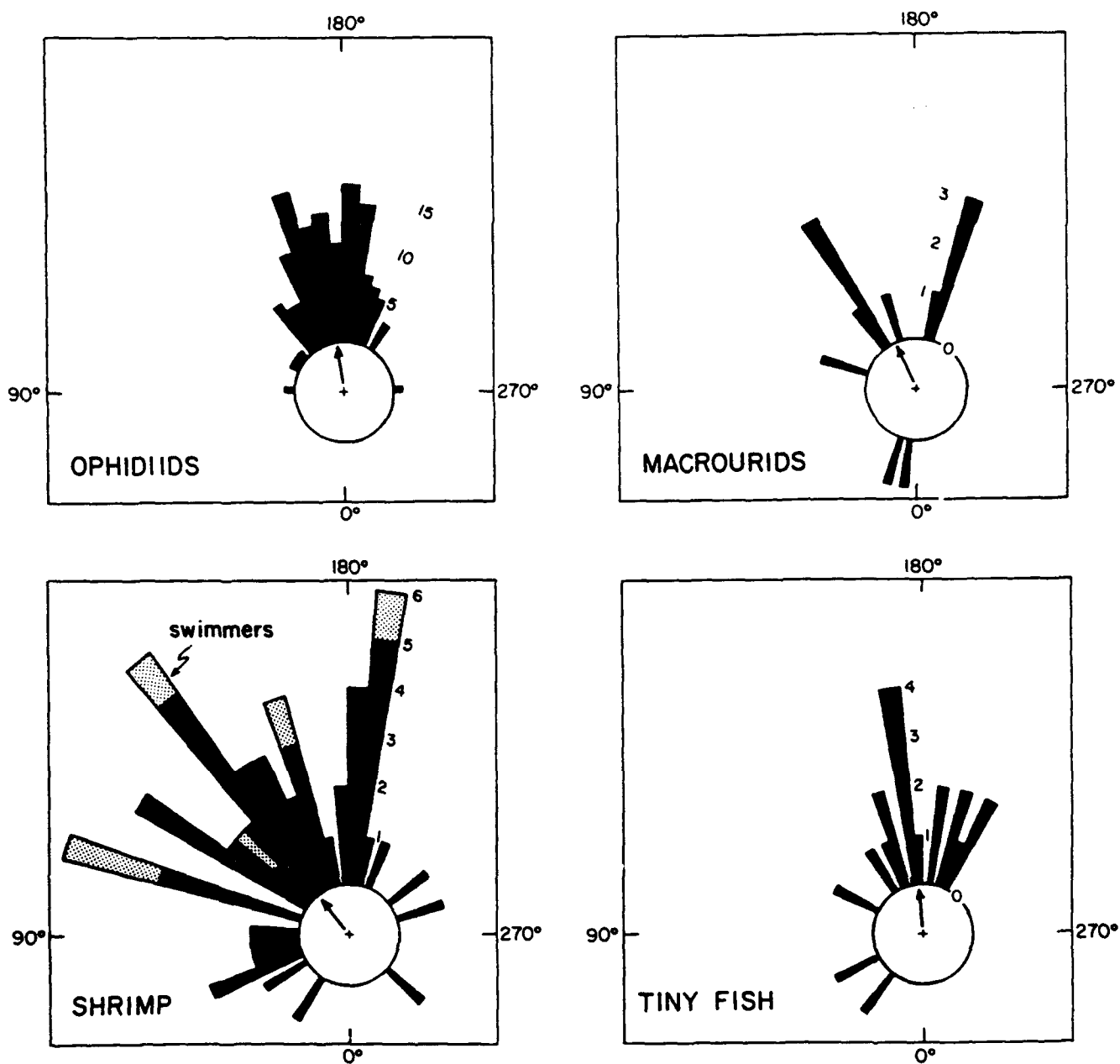


Fig. 3. Polar plots of the orientations of four nekton groups at a baited camera located at 5000 meters on Hatteras Abyssal Plain. Numbers represent number of observations; arrow indicates mean orientation angle.





**RADIONUCLIDE CONCENTRATIONS IN COMMERCIAL DEEP-SEA FISHES  
AND INTERTIDAL MUSSELS FROM THE VICINITY OF THE  
FARALLON ISLANDS NUCLEAR WASTE DUMP SITE, CALIFORNIA**

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**ABSTRACT:**

The Farallon Islands Nuclear Waste Dump Site (FINWDS), located approximately 30 miles west of San Francisco, received low level (?) nuclear waste encapsulated in recycled 55 gal drums from ca. 1945 to 1970. At least 47,500 barrels of nuclear waste is believed to contain over 14,500 Curies of radioactivity (excluding tritium) at this site, making it one of the largest known oceanic repositories in the world. Three specific regions were assigned as targets for drum disposal, but recent data suggest a much wider distribution of waste containers. Previous reports on this site from the 1970's indicated that sediment radionuclide levels were about 2-25 times above expected background levels (relative to assumed atmospheric fallout distribution). However, recalculated values indicate that sediment radionuclides may range from background to over 1000 times expected background levels.

During several seasons in 1986/87 we collected deep-sea bottom feeding fishes (Dover sole = *Microstomus pacificus*; sablefish = *Anoplopoma fimbria*; thornyheads = *Sebastolobus* spp.) and intertidal mussels (*Mytilus californianus*) from the vicinity of the FINWDS and from comparable depths at a comparison site at Point Arena approximately 100km to the north. These species were analyzed for several radionuclides ( $^{137}\text{Cs}$ ,  $^{238}\text{Pu}$ ,  $^{239+240}\text{Pu}$  and  $^{241}\text{Am}$ ). For deep-sea fishes, only muscle filet tissue was analyzed; for intertidal mussels the entire body viscera was analyzed. Field samples were fresh frozen, then thawed, dissected, homogenized, lyophilized, and ca. 30-40g aliquots were removed. For  $^{137}\text{Cs}$ , samples were compacted inside plastic Petri dish geometries and counted for 20h by gamma-ray spectrometry using 8X8X3in NaI(Tl) scintillation detectors. For  $^{238}\text{Pu}$ ,  $^{239+240}\text{Pu}$  and  $^{241}\text{Am}$ , samples were first spiked with tracers, wet ashed, dry ashed, precipitated with  $\text{Fe}(\text{OH})_3$  and separated by solvent extraction. The extracts were subjected to electrodeposition onto stainless steel planchets and analyzed using alpha-ray spectrometry with two surface barrier detectors coupled to a 4096-multichannel analyzer.

Radionuclide concentrations in fish filets and mussels ranged from 0-120 pCi/kg wet wt, with the following typical means:  $^{137}\text{Cs}$  = 30 pCi/kg;  $^{238}\text{Pu}$  = 8 pCi/kg;  $^{239-240}\text{Pu}$  = 4 pCi/kg and  $^{241}\text{Am}$  = 40 pCi/kg. These values were converted to projected per capita annual radionuclide intake from consumption of these species, yielding the following typical values: 0.1-1.0 mrem/yr for  $^{137}\text{Cs}$ ,  $^{238}\text{Pu}$  and  $^{239+240}\text{Pu}$ . For  $^{241}\text{Am}$ , projected mrem intake were highest for Dover sole, sablefish and some thornyheads ranging from 1-10 mrem/yr. Calculated dose equivalents derived from the highest of these values would be expected to yield a value of approximately 1 mrem/yr, with dose to the bone averaging ca. 0.2 mrem/yr. Ratios of the radionuclides  $^{238}\text{Pu}$ : $^{239-240}\text{Pu}$  generally ranged from ca. 2-6, considerably higher than the predicted 0.05-0.08 ratio typical for atmospheric fallout sources. Ratios for the radionuclides  $^{241}\text{Am}$ : $^{239-240}\text{Pu}$  generally ranged from ca. 8-29, also much higher than the < 0.32 predicted for atmospheric fallout sources.

## INTRODUCTION:

Between 1946 and 1970 the Farallon Islands Nuclear Waste Dump Site (FINWDS) received at least 47,500 barrels of radioactive waste (reportedly low-level radioactive waste, but see Davis (1980a, b) for an alternative opinion), with a radionuclide inventory of at least 14,500 Ci, excluding tritium (Noshkin et al., 1978). Most of the radioactive wastes deposited at the FINWDS are believed to be contained in recycled 55-gallon 16-gauge steel drums, deposited either individually or in clusters of barrels at three primary sites (at approximately 100m, 900m, and 1800m depths; Figure 1) although the present day location of the barrels is likely more widely distributed (Karl et al., 1991). Historical records indicate that approximately 150 barrels were deposited at the 100-m site (A); 3,600 barrels at the 900-m site (B); and 44,000 barrels at the 1800-m site (C). The polygon shown in Figure 1 probably more accurately describes the actual distribution of the waste containers.

The life expectancy of these metal waste containers (to effectively contain wastes) has been calculated to be ca. 10 years in seawater, whereas the life expectancy of the concrete enclosing the radioactive wastes was calculated at ca. 30 years (Joseph - Appendix X (from Carritt, 1958); Waldichuk, 1960); but the long-lived radionuclides contained within the barrels have half-lives up to ca. 24,000 years. However, no *in-situ* testing of these barrels has been conducted under deep ocean water pressures. During the 1970's only an extremely small proportion (0.34 of 1%) of the waste barrels was examined, yet a substantial number of those barrels (25-30%) had been breached presumably from implosion (Dyer, 1976). Because of their relatively short life expectancy, even intact barrels may now be reaching their functional lifespan.

Only one barrel from the FINWDS has been recovered (in 1977) from the 900m disposal site (Walden, 1987) and the condition of the barrel and contents analyzed (Colombo & Kendig, 1990). No external radiation or contamination was detected on the recovered barrel; however, this barrel was chosen specifically for its appearance of relative integrity in order to reduce the potential for exposure to ship-board personnel during recovery. Precise determination of metal loss was made difficult because of insufficient knowledge of the initial thickness of the barrel. However, these results provide some estimates of corrosion rate for metal exposed to the sediment (260-year time for 50% reduction in thickness of a 1.3-mm wall) and for metal exposed to seawater (34-year time for 50% reduction in thickness of a 1.3-mm wall). In addition there were regions on the barrel that had high rates of corrosion and perforations, especially in the areas of cold work and the chimes that were exposed to seawater. These perforations of the metal barrel were determined to be caused by processes on the seaward side of the metal, not from the concrete side (Colombo & Kendig, 1990).

Because of human health concerns, the California State legislature passed Senate Bill 444 in 1983, providing a mechanism to investigate levels of radionuclides in some species of edible fishes from the FINWDS. In response to a 1985 California Department of Health Services directive by the California Scientific Advisory Committee on the Ocean Disposal of Radioactive Waste, Pursuant to Chapter 1182, Statutes of 1983, the University of California was engaged to conduct a study to monitor levels of radionuclides in marine organisms (bottom fishes and intertidal mussels) in the vicinity of the FINWDS and from a comparison site (subsequently designated as Point Arena, ca. 100 km to the north) in order to determine whether radionuclides concentrated in tissues typically consumed by the public might pose a potential threat to human health. The organisms chosen for study included: Dover sole (*Microstomus pacificus*), sablefish (*Anoplopoma fimbria*), thornyheads *Sebastolobus* spp. and the California mussel (*Mytilus californianus*). Field collections were conducted in 1986 and 1987 and subsequent radionuclide analyses continued through 1991 (Suchanek and Lagunas-Solar, 1991).

## PREVIOUS DATA ON RADIONUCLIDES IN SEDIMENTS AND BIOTA AT THE FINWDS:

From a 1974 study at the 900m site, Dyer (1976) reported sediment radionuclide concentrations ranging from 2-25 times expected background levels, although a recalculation of his data shows that for  $^{239/240}\text{Pu}$ , the actual concentrations range from background to 1064 times background (see Suchanek, 1987 and Figure 2). Another report on sediments from near the waste barrels estimated

the range of  $^{239/240}\text{Pu}$  concentrations at 8-2208 times expected background (LFE, 1979; Davis, 1980b). From a human health perspective, there is concern that commercially exploited edible deep-sea fishes (e.g. channel rockfish, Dover sole, sablefish) may be contaminated with short- and/or long-lived radionuclides (e.g. Cs, Pu, and Am) some having half-lives exceeding 24,500 yrs (i.e.  $^{238}\text{Pu}$ ).

Several species of marine organisms have been observed and/or collected directly from the immediate vicinity of the waste containers. These include invertebrates such as vaseform sponges, brittlestars, polychaetes, sea pens, squid, sea cucumbers, anemones, snails, shrimp, and crabs as well as fishes including sablefish, thornyheads, Dover sole, deep-sea sole, Pacific flatnose, Pacific rattail, Pacific sanddab, lanternfish, loosejaw, eared blacksmelt, midshipman, rock sole, hake, deep-sea smelt and twoline eelpout (Dyer, 1976; Schell & Sugai, 1980).

Some fishes and invertebrates from this region were found to contain elevated levels of radioactivity and some fishes were believed to have radionuclide concentrations of 10-8,500 times expected background levels (Schell & Sugai, 1980; Davis, 1980b). These studies have shown that the organisms with the highest radionuclide body burdens (i.e.  $>100$  pCi/kg dry wt) are: invertebrates = polychaete worms, sea cucumbers, sponges; fishes = deep-sea smelt, Dover sole, hake, lantern fish, midshipman, Pacific flatnose, rattail fish, and Pacific flatnose (Schell & Sugai, 1980).

Because mussels are efficient filter feeders, they have been used effectively in statewide, nationwide and worldwide programs to monitor pollutant levels in the natural environment. Mussels have been found to concentrate radionuclides ca. 200-300 times the level found in surrounding seawater. The Farallon Islands have also been used as one of many California sites with which to compare national pollutant levels on both coasts of the United States through the Mussel Watch Program (Goldberg et al., 1978). However, no "regular" transuranic radionuclide analyses have been performed on mussels during the California Mussel Watch program reported by Ladd et al., (1983). This is surprising since one set of data from the EPA National Mussel Watch program clearly indicates that mussel tissues from the Farallon Islands site (collected in 1976) yielded the highest levels of  $^{239/240}\text{Pu}$  and the highest levels of  $^{241}\text{Am}$  of any samples collected from both coasts of the United States (Goldberg et al., 1978).

The EPA reported mussel tissues from Southeast Farallon Island with roughly 3.3 times the mean concentration found at all other California sites pooled. Mussel samples from this site had mean dry weight radionuclide concentration levels for  $^{239/240}\text{Pu}$  of  $3.4 \pm 0.14$  pCi/kg compared with a mean of  $1.0 \pm 0.68$  pCi/kg for all other California sites pooled ( $n=19$ , range = 0.14-2.09 pCi/kg). Mean levels for  $^{241}\text{Am}$  were  $8.91 \pm 0.68$  pCi/kg compared with  $2.68 \pm 2.95$  pCi/kg for all other California sites pooled ( $n=17$ , range = 0.04 - 7.86 pCi/kg). No radionuclide analyses were performed on any specific organs or the shells of mussels (only viscera) from the Farallon Islands site.

#### ECOLOGY OF BOTTOM-FEEDING FISHES SAMPLED:

Three taxa of fishes (Dover sole, sablefish, and thornyheads) were collected because of their direct applicability to potential human health risk through consumption. The following information provides natural history data that are relevant to each of these taxa.

Dover sole (*Microstomus pacificus*) is a bottom-feeding species that generally occurs on muddy substrata and ranges from ca. 180-3900 ft depth. Its larvae have been sighted offshore to 280 miles yet it is not a widely migrating species. It has limited coastwise movement and several isolated sub-populations are believed to exist (Frey, 1971). The Dover sole does, however, undergo extensive seasonal onshore/offshore movements related to its spawning cycle. During spring through summer it typically feeds extensively in inshore waters. During November-March it moves offshore for spawning, where it produces buoyant pelagic eggs. It feeds exclusively on benthic invertebrates including: bivalves, scaphopods, sipunculids, polychaetes, echinoids, ophiuroids, gastropods, and crustaceans (Frey, 1971).

466 Sablefish (Anoplopoma fimbria) prefer soft bottom habitats like the Dover sole and are found at depths greater than 300 ft. They are not known to migrate for spawning purposes, but migration is likely important in this species, as one individual tagged in Japan was later found in the United States (Frey, 1971). It is believed that there is one sub-stock from central to southern California. Spawning generally occurs from December to April with a peak in January/February. Juveniles are known to feed on the following benthic invertebrates: copepods, amphipods, euphausiids, fish eggs, fish larvae and the larvacean Oikopleura. Subadults and adults are generally believed to feed on euphausiids, tunicates and fish (especially anchovy) (Frey, 1971).

Thornyheads (also called idiot-fish or channel rockfish) (shortspine = Sebastolobus alascanus; longspine = S. altivelis) are non-migratory deep-water species that are generally known to range from 1800-2520-ft depth although they likely occur deeper (Phillips, 1957; Frey, 1971).

California mussels (Mytilus californianus) are moderately long-lived (likely 20-50yrs) and intertidal specimens were also chosen for this study because (1) they are very efficient filter feeding bioaccumulators and have been used extensively as indicator species for the presence of heavy metals and/or radionuclides, and (2) there are some previous data to indicate that M. californianus from Southeast Farallon Island had significantly elevated radionuclide concentrations (Goldberg et al., 1978).

## METHODS:

### SELECTION OF A REFERENCE SITE:

The location of the reference site was chosen on the basis of the best available data on water movement in this region in order to provide the least potential for contamination from the FINWDS. Although the directionality and speed of currents in the Gulf of the Farallons is variable seasonally, the largest annual mass movement of water from the vicinity of the FINWDS is southward (Conomos et al., 1971; Conomos, 1975; Conomos and Peterson, 1977). Surface currents from FINWDS generally show significant northward and/or southward movement along the coast, whereas bottom currents are more complex (Conomos et al., 1971; Conomos, 1975; D. Lindberg, pers. comm., unpublished data; Dyer, 1976; Conomos & Peterson, 1977). One current meter placed on the bottom at the 1829m site during 1975 showed essentially northward bottom current movement at speeds of ca. 1.17 km/day (Dyer, 1976; Crabbs, 1983). Bottom currents moving in this direction would likely transport particles toward the vicinity of Cordell Bank, a region used extensively for commercial and sport fisheries. However, another bottom current study (using seabed drifters released in the Gulf of the Farallons) indicated consistent eastward movement of bottom currents at speeds of at least 0.5 km/day (Conomos et al., 1970, 1971; McCulloch et al., 1970; Conomos & Peterson, 1977). Significantly, these currents move particles eastward along the sea floor, with final destinations in San Francisco Bay and San Pablo Bay. Therefore, a reference location was chosen at Point Arena, ca. 100km north of the FINWDS.

### FIELD COLLECTIONS:

Fish and mussel collections were made during three periods: (1) December 1986/January 1987, (2) May/June 1987 and (3) August/September 1987.

Deep-sea fishes were collected using commercial fishing trawlers (typical trawling period = 1-4 hours) in the vicinity of the 900m depth FINWDS dump site and from a comparable depth off the Point Arena coast. Individuals of each target species were selected haphazardly (but included a range of sizes) by hand from hauls that typically contained several tons of material. After collection, fish were immediately iced and placed in polyethylene plastic bags on board the vessel (1-2 day holding time). After return to port they were preserved by freezing (-20°C) until dissection.

California mussels were collected intertidally by hand from Southeast Farallon Island and from a reference intertidal site at the base of the Point Arena Lighthouse. Mussels were placed in

polyethylene bags and held on ice during transit; they were subsequently frozen whole at  $-20^{\circ}\text{C}$  until dissections were performed.

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#### SAMPLE PREPARATION: DISSECTION. HOMOGENIZATION. LYOPHILIZATION AND PRESERVATION:

Fishes were sexed, weighed and measured for total length, fork length, and standard length. The following organs/tissues were dissected and weighed for each fish: filets (muscle tissue only), skin, liver, gonads, kidney, otoliths, and G.I. tract. Only filets were analyzed for radionuclides. Other organs/tissues (as well as the remainder of the filets and carcasses, including the bones) remain archived in a frozen state ( $-20^{\circ}\text{C}$ ) at UC Davis.

Mussels were dissected using methods from Goldberg et al. (1983). Mussels were sexed and shells measured for total length, width and height. Wet weight of the shell, byssal threads (if still attached) and composite viscera were recorded. The remaining tissues, shells and byssal threads also remain archived in a frozen state.

Tissues from both fishes and mussels (ca. 750-1200g wet wt) were placed in a stainless steel Waring Blender and homogenized. Double-distilled deionized water was added to some samples in order to achieve a smooth consistency in the homogenate. This homogenate slurry was then poured into aluminum foil "pans", flash frozen (at  $-50^{\circ}\text{C}$ ) and lyophilized for ca. 4-5 days in a Vertis™ freeze-drier to form a thick wafer of tissue. Water removed from the tissues which had accumulated in the freeze-drier during the lyophilization process was analyzed and no above-background alpha- or gamma-ray activity was detected in these samples. Wet to freeze-dried weight ratio approximated 10:1, but varied among samples.

Samples of lyophilized fish and mussel tissues were compressed into a geometry suitable for gamma-ray spectrometry. This was accomplished by compressing approximately 30-40g of lyophilized tissue into a three-dimensional disk geometry (a standard 9cm diameter x 1.3cm thick Petri dish), forming a sample of relatively uniform thickness and density. In order to prevent decomposition of samples during a long storage period, the samples were sterilized (3.5-4.0 Mrads/sample) using the U.C. Davis Cobalt-60 Mark-II Irradiator.

#### GAMMA SPECTROMETRY:

The analyses of  $^{137}\text{Cs}$  radioactivities using the 662-keV (85%) gamma-ray were conducted with standard gamma-ray spectrometry methods using two 8x8 in. (3-in thick) NaI(Tl) scintillation detectors, one on either side of the Petri dish to accomplish  $2\pi$  geometry. No radiochemistry was conducted on samples being analyzed for  $^{137}\text{Cs}$ . Each lyophilized sample was counted for 20 h. The total gross count was corrected for background (including  $^{40}\text{K}$  contribution), probability per decay, and detector efficiency.

The  $^{137}\text{Cs}$  radioassays were conducted at the U.C. Davis Laboratory for Energy-Related Health Research (LEHR) using detectors equipped with a multi-channel analyzer system. The gross  $^{137}\text{Cs}$  in the 662-keV photopeak and the corresponding background (BKG) counts were used to calculate the net  $^{137}\text{Cs}$  count rate using the linear-sum method. The  $^{137}\text{Cs}$  minimum detectable activity (MDA) for these systems was  $> 5$  pCi/sample with ca. 30% total counting efficiency. The total uncertainty of the  $^{137}\text{Cs}$  radioassays equaled  $\pm 34\%$  (2 d; 95.4% confidence level) and was determined from the individual uncertainties (errors) using the root-mean-square method. The individual uncertainties, expressed in percentage (%) included: sample dry weight ( $\pm 1\%$ ); count rate uncertainties ( $\pm 10\%$  [max.]); timing errors ( $\pm 1\%$ ); detector efficiencies ( $\pm 3\%$ ); source-detector geometry variations ( $\pm 3\%$ ); photopeak fitting errors ( $\pm 8\%$ ); secondary references ( $\pm 5\%$ );  $^{40}\text{K}$  background corrections ( $\pm 8\%$ ); and natural background subtractions ( $\pm 5\%$ ). Due to the extremely low count rate of the samples, no other corrections (i.e. detector dead-time, pile-up, summing, etc.) were needed.

468 The efficiency of the NaI(Tl) detector system was determined with several  $^{137}\text{Cs}$  "disc-geometry" standards prepared in an agar matrix by dilution from a N.S.I.T. (ex-NBS)  $^{137}\text{Cs}$  reference solution standard. These secondary  $^{137}\text{Cs}$  reference standards were prepared in the 10-100 pCi range (i.e. 10-; 30-; 50-; 60-; and 100-pCi/std) and used periodically to calibrate the NaI(Tl) counting systems.

#### RADIOCHEMICAL PREPARATION BEFORE ALPHA SPECTROMETRY:

The alpha radioassays were determined only after subjecting the lyophilized samples to extensive radiochemical processing using the radiotracer method with  $^{243}\text{Am}$  ( $7.4 \times 10^3$  yr;  $\alpha$ 's 5.28 MeV [88%], 5.23 MeV [11%]; measured as 5.27 MeV [100%]) and  $^{242}\text{Pu}$  ( $3.8 \times 10^5$  yr;  $\alpha$ 's 4.90 MeV [74%], 4.86 MeV [26%]; measured as 4.89 MeV [100%]) as radiotracer spikes, and electroplating (fixing) the alpha radioactivities on the surface of a 1-in. o.d. stainless steel planchet. The radiochemical methods used to separate Am and Pu from fish and mussel samples were adapted from Singh *et al.* (1982) [from the University of Utah] and from Boyd *et al.* (1979) and are outlined in the methodology flow diagram in Figure 3 (Lagunas Solar *et al.*, in preparation).

Spiking and Wet Ashing of Lyophilized Samples: Approximately 40g of each lyophilized tissue sample was first reconstituted with doubly distilled water to a manageable consistency and spiked with ca. 5 dpm of  $^{243}\text{Am}$  and 5 dpm of  $^{242}\text{Pu}$  from NSIT referenced solution. Samples were wet ashed using concentrated  $\text{HNO}_3$  at 70°C. Ashing was continued in a muffle furnace at increasing temperatures up to 500°C over for up to ca. 100 hrs. The reduction of fresh (lyophilized) sample to ash weight was approximately 10:1.

#### Precipitation, Separation and Extraction:

Am, Th, Pu, U were co-precipitated with  $\text{Fe}(\text{OH})_3$ . Pu, U, and Am were separated by solvent extraction using Tri-lauryl amine (TLA) (also known as Tridodecylamine). Pu and U were extracted using 1.0M  $\text{NaNO}_2$  and the Pu was further separated using 0.05M  $\text{NH}_4\text{I}$ . Pu was further purified by solvent extraction using ether. Pu was then electrodeposited on a stainless steel planchette using methodologies adapted from Singh *et al.* (1982).

Am was separated using ether solvent extraction and anion exchange procedures using AG/MP-1 anion exchange resin. Am was then electrodeposited onto a stainless steel planchette.

#### ALPHA SPECTROMETRY:

Alpha-ray radioassays (for Pu and Am) were performed at the U.C. Davis Crocker Nuclear Laboratory using alpha-ray spectrometry with two surface barrier detectors coupled to a 4096-multichannel analyzer.  $^{238}\text{Pu}$  (87.74 yr),  $^{239+240}\text{Pu}$  (24,000 yr; 6,600 yr) and  $^{241}\text{Am}$  (432 yr) were measured using their characteristic alpha-ray emissions<sup>(1)</sup>. Both surface barrier detectors were calibrated using alpha standards referenced to N.S.I.T. The standardization was conducted using a fixed "close geometry" for counting samples supported onto a flat disc (planchet). The detector efficiencies ranged from 37.2% (detector A) to 35.6% (detector B).

The total uncertainty of the alpha radioassays was variable and largely dominated (as expected) by counting statistics. The total uncertainties for the alpha radioassays were determined from estimates of the maximum individual uncertainties (errors) using the root-mean-square method. The individual uncertainties included: Fixed Sources = sample dry weight ( $\pm 1\%$ ); timing errors ( $\pm 1\%$ ); detector

(1)  $^{241}\text{Am}$  (5.49 MeV [86%], 5.44 MeV [13%]; was measured as 5.48 MeV [100%]);  $^{238}\text{Pu}$  (5.50 MeV [71.1%], 5.46 MeV [27.8%], was measured as 5.49 MeV [100%]); and  $^{239, 240}\text{Pu}$  (5.16 MeV [73.3%], 5.15 MeV [15.1%], 5.10 MeV [11.5%]; and 5.16 MeV [75.5%] and 5.12 MeV [24.4%]; were measured combined as 5.15 MeV [100%]).

efficiencies ( $\pm 3\%$ ); source-detector geometry variations ( $\pm 3\%$ ); sample self-absorption errors (estimated) ( $\pm 15\%$ ); photopeak fitting errors ( $\pm 8\%$ ); secondary references ( $\pm 5\%$ ); natural background subtractions ( $\pm 5\%$ ); Variable (Sample Dependent) Sources = radiotracer count rate uncertainties; unknown count rate uncertainties; and radiochemistry yield (spike recovery) errors. No detector dead-time corrections were needed due to the extremely low count rate of the samples. The alpha counting protocol included periodic background counting, alpha energy resolution tests, and detector efficiency measurements as part of the system's quality assurance testing.

## **RESULTS:**

### **RADIOCHEMICAL YIELDS:**

Radiochemical yields were based on the recovery of a radiotracer spike (i.e.  $^{243}\text{Am}$  for Am radionuclides and  $^{242}\text{Pu}$  for Pu radionuclides) which was added at the beginning of the radiochemical process (see Figure 3).

#### **Average Radiochemical Yields:**

The Pu radiochemical yields averaged 19% for Dover sole samples (range 2-58%,  $n=10$ ); 14% for sablefish samples (range 3-28%,  $n=9$ ); 27% for thornyhead samples (range 11-65%,  $n=6$ ); and 17% for mussel samples (range 16.6-17.4%,  $n=2$ ). For Am, the radiochemical yields averaged 13% (range 1.4-40%,  $n=10$ ) for Dover sole samples, 9% for sablefish samples (range 1-35%,  $n=6$ ), 9% for thornyhead samples (range 2-17%,  $n=5$ ) and 21% for mussels (range 6-51%,  $n=3$ ). The latter is the only instance in which Am exceeded the Pu recovery yields. However, the size of this sample ( $n=3$  for Am; and  $n=2$  for Pu) does not provide a high degree of significance to these data. With the above results, the average radiochemical yields for Pu and Am in this study (including the EPA samples) were 20% and 14%, respectively.

#### **Electroplating Yields:**

Average electroplating yields for Pu were measured at 61% while Am yields were 40%. Analysis of Pu samples performed at UC Davis were in good agreement with comparable analyses conducted at the University of Utah (see Methods). In analyzing the results, a trend was clearly established indicating that Pu radiochemical yields were consistently higher than those for Am radiochemistry.

### **RADIONUCLIDE CONCENTRATIONS:**

Results of  $^{137}\text{Cs}$  radioassays were consistently low and near the minimum detectable activity limits of the radiation detection instrumentation. Because these samples were radioassayed with minimal preparation and manipulations (i.e. no radiochemistry), the results showed a high-degree of consistency with only a few variations. Therefore, a comparison between the different species analyzed, and between the different locations, can be accomplished on the basis of average results.

Statistically there are no significant differences in  $^{137}\text{Cs}$  content among the samples analyzed individually or in groups. It is interesting to note, however, that the control (Point Arena) mussel samples (mean =  $189 \pm 84$  pCi/kg dw) and the total (Point Arena and Farallon) thornyhead samples (mean =  $124 \pm 4$  pCi/kg dw) were the extreme (high and low) mean values found.

The results of the alpha radioassays showed a high degree of variability in all of the sample groups. This variability most likely reflects, to a certain extent, the various sources of uncertainties present in the radiochemical procedures described above, especially because of the high fat content exhibited in most of the fish tissues.

In order to determine the specific radionuclide burden for each treatment group of fishes and mussels, the total activity (pCi/kg freeze-dried wt) for each radionuclide was divided by the wet/dry ratio (a function of the amount of water removed during lyophilization) to obtain the extrapolated radionuclide

burden for wet weight tissues (pCi/kg wet weight). For the purposes of this proceedings volume, these results are provided in graphical form in Figure 4A.

Radionuclide concentrations for wet weight tissues were then converted into a projected annual radionuclide intake for each sample type by utilizing a multiplier of 25 kg/yr, given by the World Health Organization (1988) as a predicted consumption value for a Mediterranean diet. From the total radionuclide intake, a projected millirem dosage per caput was calculated by utilizing conversion factors for each radionuclide provided by the National Council on Radiation Protection and Measurements (NCRP 1987). Each annual radionuclide consumption value (pCi/kg wet weight) was multiplied by the following conversion factors to obtain the projected millirem dosage per caput:  $^{137}\text{Cesium} = 5.0 \times 10^{-5}$ ,  $^{238}\text{Plutonium} = 5.6 \times 10^{-4}$ ,  $^{239/240}\text{Plutonium} = 6.3 \times 10^{-4}$ ,  $^{241}\text{Americium} = 6.3 \times 10^{-3}$ . These results are provided in graphical form in Figure 4B.

The NCRP (1987) also provides recommendation limits for exposure to non-medical man-made sources of radioactivity. Non-occupational exposure limits (for members of the general public) are categorized into three ranges: (1) Infrequent Exposure: Annual effective dose equivalent not to exceed 500 mrem; (2) Continuous (Frequent) Exposure: Annual effective dose equivalent not to exceed 100 mrem; (3) Negligible Individual Risk Level (NIRL): Annual effective dose equivalent below 1 mrem. According to the NCRP (1987) the Negligible Individual Risk Level (NIRL) is defined as: "a level of average annual excess risk of fatal health effects attributable to irradiation, below which further effort to reduce radiation exposure to the individual is unwarranted."

It is apparent that all of the projected values for  $^{137}\text{Cs}$ ,  $^{238}\text{Pu}$  and  $^{239/240}\text{Pu}$  in Figure 4B fall below the NIRL. For  $^{241}\text{Am}$  the projections are more variable. The mean values for the pooled Dover sole and pooled sablefish samples all exceed the NIRL, with projected annual dosages ranging from ca. 0.2-18.5 mrem/yr. For thornyheads most values are clustered fairly close to the NIRL. However, even if the highest levels of projected mrem dosage for  $^{241}\text{Am}$  are applied, calculation of a "committed effective dose equivalent" yields approximately 1 mrem per year, with average doses for bone uptake equal to 0.2 mrem/yr.

Utilizing conservative t-tests in comparing values from the FINWDS versus those from Point Arena, our results indicate that there are no statistically significant differences in (1) radionuclide concentrations and (2) projected mrem dosage intake for humans, between those fishes from the FINWDS and the Point Arena reference site. However, these fishes are all quite mobile, with seasonal or annual migration patterns moving on/off shore and/or along the coast with some migrating as far as Japan (see earlier discussion under natural history of fish species), so it is not surprising that no large differences were found.

#### Radionuclide Ratios:

Measured alpha radioactivity ratios are not in agreement with ratios predicted as a consequence of atmospheric fallout. Dyer (1976) indicates that predicted ratios for the radionuclides  $^{238}\text{Pu}$ : $^{239-240}\text{Pu}$  should range from ca. 0.05-0.08, although this was indicated for sediments. Our ratios for fish muscle tissues ranged from 0.17-8.40 and for a single mussel viscera sample was 2.52 (see Table 1). Our results for the ratios of the radionuclides  $^{241}\text{Am}$ : $^{239-240}\text{Pu}$ , generally expected to be lower than 0.32 for tissues (Goldberg et al. 1978), ranged from 0.15-309.83 and represented considerably higher values than those reported elsewhere.

## DISCUSSION AND CONCLUSIONS

The precision of the alpha radioassays for Am and Pu radioactivities were verified through the use of an inter-laboratory comparison. The effect of enhanced resolution in the data analysis of alpha spectra did not have a significant effect on the magnitude of Am and Pu radioassays reported previously. The revision of the radiochemical methods and yields suggests several reasons for the efficiencies lower than the existing expectations for this kind of analyses. No evidence of a scientific validation of the



precision of the radiotracer method was found for this kind of analysis, in particular for Am radiochemistry in fish samples.

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Primarily because of budgetary constraints, not all samples collected were analyzed for radionuclides. Therefore, we were only able to use a subset of fish and mussel samples to draw conclusions regarding (1) potential significant differences between the Farallon Islands Nuclear Waste Dump Site (FINWDS) and the designated comparison site off the Point Arena coast, and (2) the projected millirem dosage that would be expected from human consumption of these species.

#### Radionuclide Concentrations and Ratios:

Unfortunately no literature on  $^{241}\text{Am}$  levels in marine fishes was found to which radionuclide burdens could be compared, so it is difficult to determine whether these represent unusually high values for oceanic fish species or not. However, some considerations should be given to the levels of projected millirem dosages provided by these results. First, these exposures would occur by consumption, not by external exposure, which is believed to be the route most likely considered by the NCRP (1987) recommendations. As such, these radionuclide emitters would likely remain within the biological system for a longer period of time, until metabolic release. How this might influence the "effective" annual dosage is beyond the scope of this project. Second, this source of exposure (i.e. from consuming fish with potentially elevated levels of  $^{241}\text{Am}$ ) is derived from a single source only. Humans are exposed to many other sources of natural and anthropogenic radioactivity annually. Kathren (1984) provides estimates of dose equivalent mean values for natural versus anthropogenic sources of radioactivity, but only ca. 25-35 mrem/yr are internal whole body sources. From the total estimated mean dose equivalent of 355 mrem/yr, consumption of a steady diet of these fish with elevated  $^{241}\text{Am}$  levels could increase this value by approximately 10% to ca. 365-370 mrem/yr.

Consideration should also be given to the type of tissues being analyzed. For fish, these analyses (as specified objectives of this research) were conducted on muscle tissue (fillets) only. Because many ethnic groups consume other fish tissues, including skin and bone, the use of only fish fillet tissues may be misleading in that it provides an incomplete picture to evaluate questions involving potential human health risk. Furthermore, earlier literature indicates that skin tissue often contains considerably higher radionuclide concentrations than does muscle tissue (see Schell & Sugai, 1980).

Our derived ratios for both  $^{238}\text{Pu}:^{239-240}\text{Pu}$  and  $^{241}\text{Am}:^{239-240}\text{Pu}$  exhibited levels that are elevated from those predicted to derive from atmospheric fallout alone (Table 1). This is not surprising given that the FINWDS represents a significant source of waste radionuclides (see Dyer, 1976; Suchanek, 1987; Suchanek and Lagunas-Solar, 1991). Anomalous ratios have also been noted in the vicinity of nuclear power plants along the east and west coast of the United States (Goldberg et al., 1978).

#### Other Studies Near the Farallon Islands Nuclear Waste Dump Site:

Other recent studies also show an abnormally high incidence (up to 38%) of chromatophoromas and related hyperplastic lesions in Pacific rockfish (*Sebastes* spp.) just north of the FINWDS at Cordell Bank, with the frequency of raised neoplasms and lesions for at least one species (*S. flavidus*) increasing over the 5 year period from 1985-1990 (Okihiro et al., 1993). Whether these symptoms are related to the possible exposure to a radioactive carcinogen in the regional environment has yet to be determined, although other recent collections from the vicinity of the FINWDS show a relatively high incidence of lesions (Okihiro, pers. comm.).

With renewed interest in the potential for radionuclide contamination and human health risk from the FINWDS, congressional funds were appropriated in 1991 from both NOAA and EPA sources (approximately \$900,000 total) to reevaluate the status of the dump site. The first stage in this reevaluation is an inventory search of records to determine as accurately as possible: (1) the contents of the waste barrels; (2) the radionuclide inventory, originally estimated to be ca. 14,500 Ci (exclusive of tritium,  $^3\text{H}$ ); and (3) the location of sites where barrels were believed to be deposited. While this is clearly a useful exercise, regardless of what a paper record does or does not provide in terms of a radionuclide inventory, the essential elements needed to determine the current level of anthropogenic radionuclide contamination at the FINWDS and any subsequent potential human health

472 risk are a current survey of the levels of radionuclides in the physical and biological components at the site today. Our study provided information primarily for radionuclide concentrations in highly mobile fishes. It is recommended that future studies at this site include analysis of radionuclide concentrations in sediments, near-bottom waters and organisms that are permanent benthic residents benthic invertebrates.

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Figure 1. Location map for Farallon Islands Nuclear Waste Dump Site.

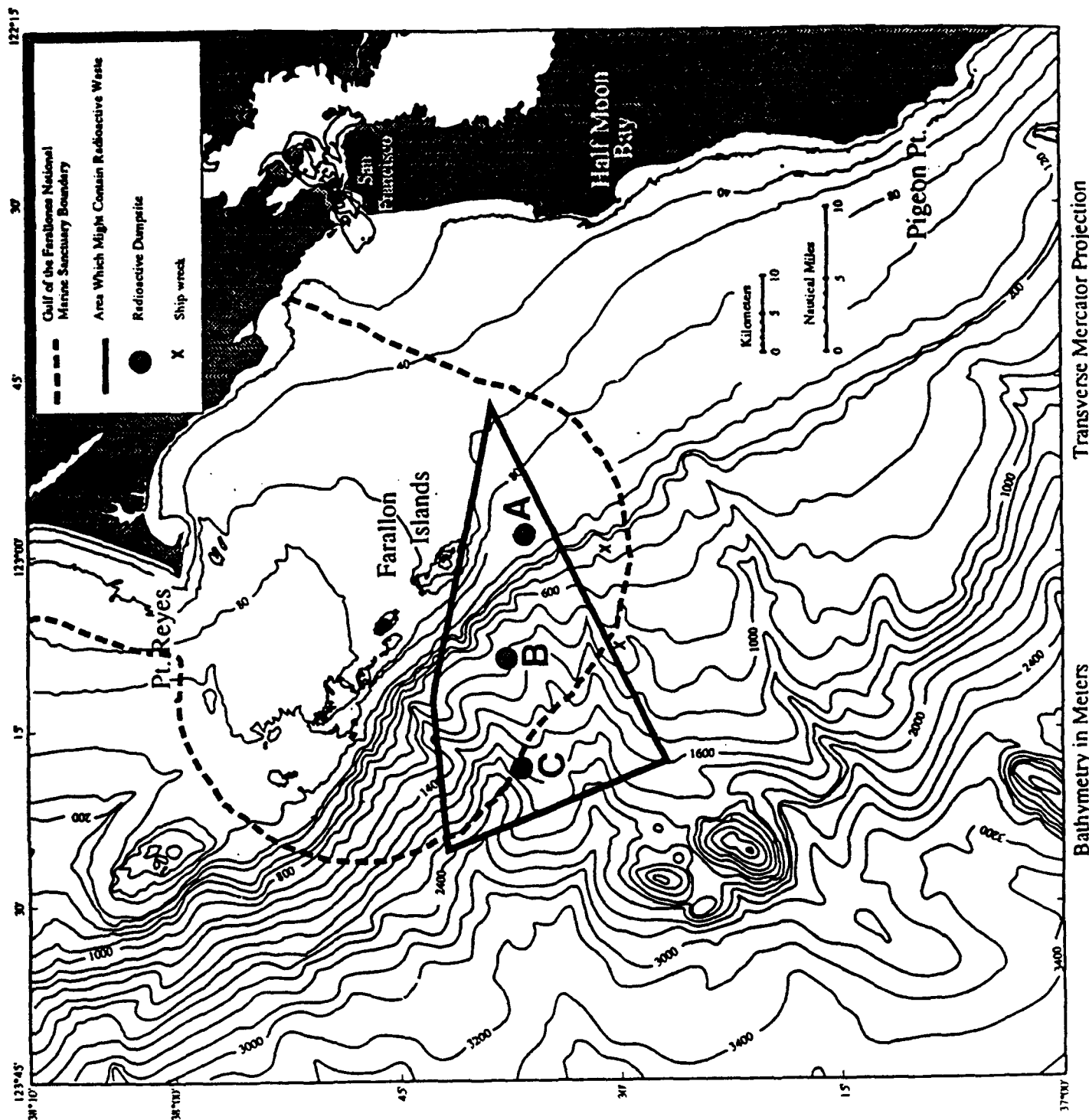


Figure 2. Data from Dyer (1976) for  $^{239,240}\text{Pu}$  levels in sediments at the 900-m site of the Farallon Islands Nuclear Waste Dump Site. Shaded regions identify Dyer's reported results. Black bars represent recalculated values from Dyer's Table 2 (1976) as identified by Suchanek (1987).

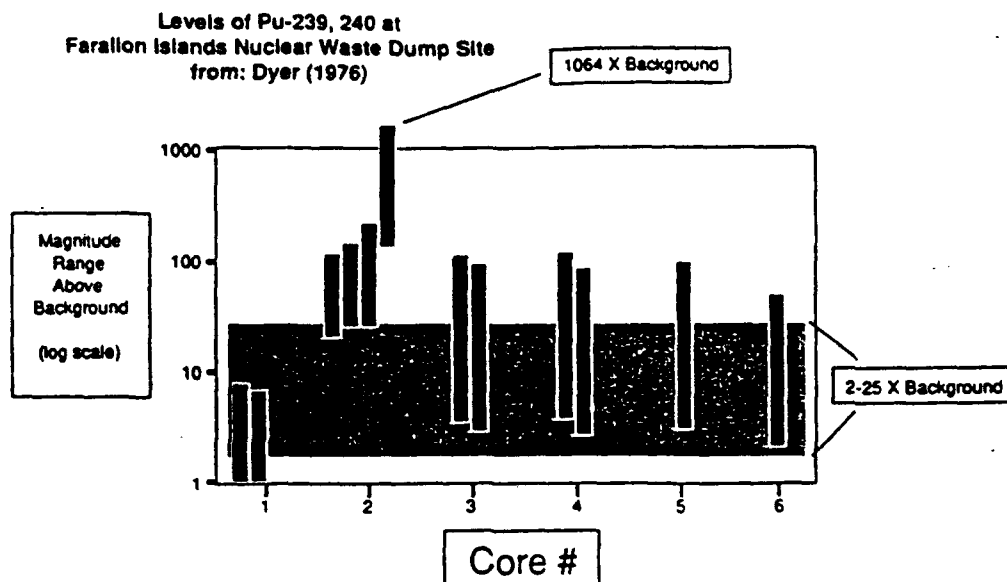


Figure 3. Schematic flow diagram for Pu/Am radiochemical analyses.

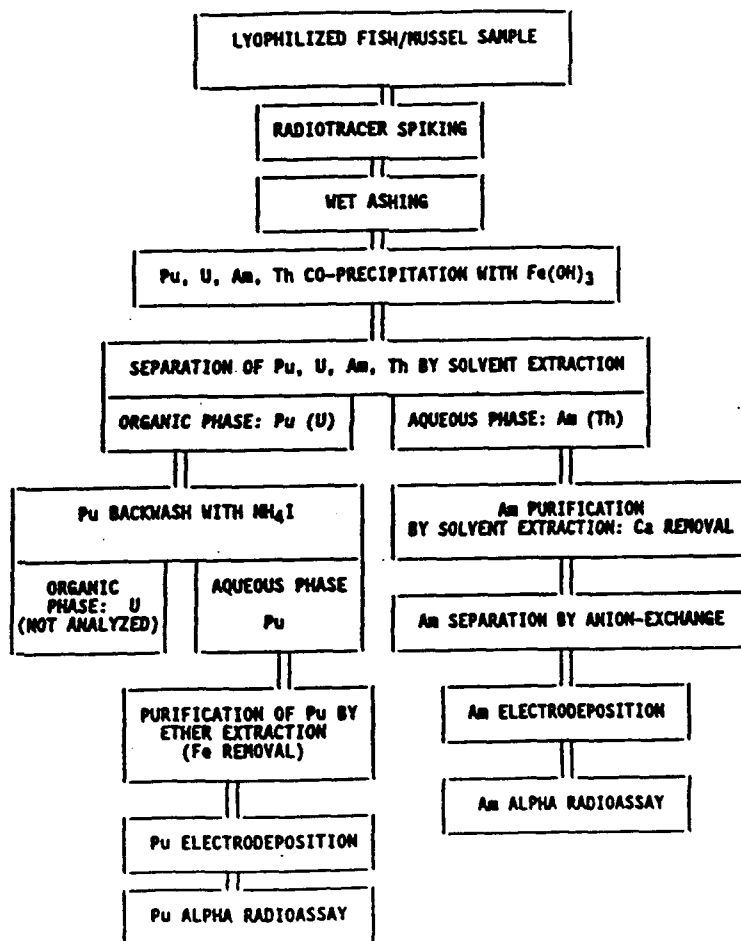


Figure 4A. Radionuclide concentrations in deep-sea fishes and mussels from the vicinity of the Farallon Islands Nuclear Waste Dump Site.

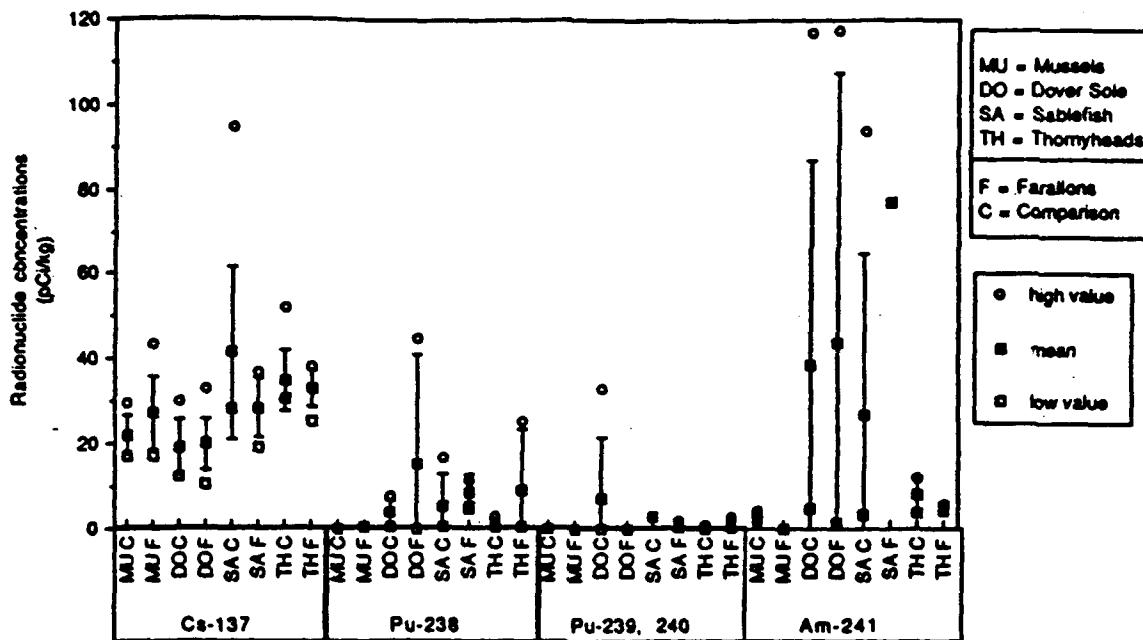


Figure 4B. Millirem intake per year from typical consumption of deep-sea fishes or mussels from the vicinity of the Farallon Islands Nuclear Waste Dump Site.

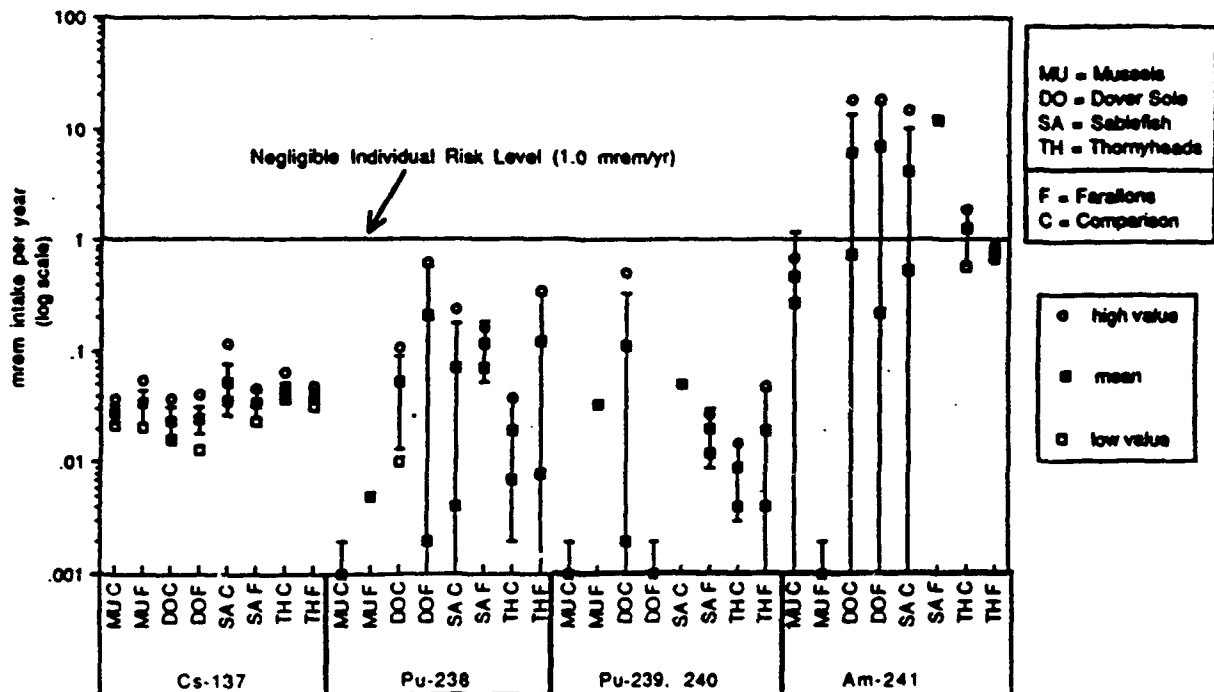


Table 1. Radionuclide Ratios for biological samples from the Farallons Dump Site and Point Arena Comparison Site.

FARALLONS DUMP SITE	Pu-238/Pu-239,240				Am-241/Pu-239,240			
	mean +/-	stand. dev.	range	n	mean +/-	stand. dev.	range	n
Mytilus (mussel)	2.52 +/-	na	2.52	1	no data		nd	nd
Dover Sole	no data		nd	nd	no data		nd	nd
Sablefish	5.53 +/-	3.58	3.00-8.06	2	no data		nd	nd
Thornyheads	4.47 +/-	3.44	2.00-8.40	3	18.50 +/-	2.12	17.00-20.00	2
<b>POINT ARENA COMPARISON SITE</b>								
Mytilus (mussel)	no data		nd	nd	no data		nd	nd
Dover Sole	3.73 +/-	2.46	0.17-6.25	6	77.45 +/-	117.97	0.15-309.83	6
Sablefish	5.36 +/-	na	5.36	1	29.82 +/-	na	29.82	1
Thornyheads	2.25 +/-	0.43	2.00-2.75	3	17.58 +/-	10.37	8.75-29.00	3





Radiological Assessment for  
the Dumping of Radioactive  
Wastes in the Oceans

Conference on Radioactivity  
and Environmental Security in  
the Oceans: New Research and  
Policy Priorities in the Arctic  
and North Atlantic

W.L. Templeton

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## **Radiological Assessment for the Dumping of Radioactive Wastes in the Oceans**

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### **Background and History**

The purpose of this paper is twofold. First, the paper provides a historical review of international activities regarding radiation in the sea. Secondly, the paper provides some recommendations for future research needed for realistic dose assessment of the present and potential impact of ocean disposal and dumping operations.

### **Introduction**

Over the last three decades or so, a number of international meetings have been convened to treat the specific problem of radioactive waste disposal into the oceans. The first of these meetings was held in 1958 at the United Nations Conference on the Law of the Sea. Immediately following, the International Atomic Energy Agency (IAEA), in the Brynielsson Report, recommended measures for ensuring that disposal of radioactive waste into the sea would not result in unacceptable hazards to man (IAEA 1961). Since that time, major changes have occurred in the philosophy and recommendations of the International Commission on Radiological Protection that are crucial to the assessments of impacts arising from this practice. Knowledge of oceanographic processes has improved markedly, providing better understanding of the physical transport process and of the pathways by which radionuclides are transported from marine dumping and disposal sites back to man. Finally, radioecology has developed to the stage where predictions of radionuclide cycling pathways and rates are possible. The IAEA report of 1961 was revised in 1983 (IAEA 1983). The IAEA has published many documents (Safety Series and Technical Documents) covering relevant areas such as oceanographic models, bioaccumulation factors, sediment distribution coefficients, and effects of ionizing radiation on organisms. The IAEA has also convened several symposia dealing with subjects related to sea disposal, such as radionuclide cycling in the marine environment, radioactive waste management, radiological and

environmental protection, and environmental surveillance. Additionally, a large number of papers on these subjects have been published in the scientific literature.

The number of international and regional conventions has also increased over the same time period. One resolution of the United Nations (UN) Conference on the Human Environment, held in Stockholm in 1972, provided the stimulus for the formulation of the Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter (London Dumping Convention, 1972) which entered into force in 1975. The Convention binds participating nations to take all practical steps to prevent the pollution of the oceans through dumping of wastes which may create hazards to human health, harm living resources and marine life, damage amenities, or interfere with other legitimate use of the seas. It should be noted that the disposal of liquid wastes in coastal waters is not covered by the convention (International Maritime Organization [IMO] 1982).

The London Convention 1972 entrusts the IAEA with specific responsibilities for dumping of radioactive wastes at sea, and for making recommendations for radiological dose assessment and surveillance. In 1974, the IAEA established a Provisional Definition of High Level Wastes or other radioactive matter unsuitable for dumping at sea, and for dumping other material at sea. In this document, the IAEA made recommendations that the Contracting Parties should take fully into account in issuing permits for dumping other radioactive wastes or radioactive matter at sea. The IAEA recommendations were adopted by the London Convention 1972 in 1976, revised in 1978, and again revised in 1985 (IAEA 1986). The material deemed by the London Convention as unsuitable for dumping at sea includes irradiated reactor fuel, liquid wastes from the first extraction cycle of reprocessing irradiated fuel and solidified forms of such wastes, and any other wastes of concentrations exceeding specified quantities of alpha emitters; beta/gamma emitters with half-lives greater than one year; and tritium and beta/gamma emitters with half lives of one year or less. All other wastes with activity concentrations less than those specified shall not be dumped except in accordance with the provisions of the Convention and Recommendations.

The Derivation of the Quantitative Definition of Wastes Unsuitable for Dumping at Sea is given in Appendix 1 of IAEA Safety Series No. 78 (1986).

The maximum dumping rate into a single ocean basin of volume of at least  $10^{17} \text{ m}^3$  shall not exceed  $10^8 \text{ kg}$  per year. While virtually all materials contain some radionuclides, it is not the intention that all materials be treated as radioactive when considering their suitability for dumping at sea. For example, sewage sludge, dredge spoils, fly ash, agricultural wastes, construction materials, vessels which are not nuclear powered, artificial reef building materials, and other such materials that have not been contaminated with radionuclides of anthropogenic origin (except global fallout from nuclear weapons testing) are not considered to be radioactive for the purposes of sea disposal.

Additional requirements must be met by the appropriate national authorities in selecting of a site for dumping of packaged wastes. Two of the most important are:

- Dumping shall be restricted to those areas of the oceans between latitudes  $50^\circ\text{N}$  and  $50^\circ\text{S}$ . The area shall have an average depth greater than 4000 meters.
- The site should be located clear of the continental margin and open sea islands and not in marginal or inland seas.

#### Naturally-Occurring Radionuclides in the Oceans

Naturally-occurring nuclides can be measured throughout the ocean environment and are an important source of radiation for organisms, as is the case of naturally-occurring nuclides in the terrestrial environment. Primordial nuclides include those of the uranium and thorium chains such as radium-226, lead-210, polonium-210, and potassium-40. The cosmogenic nuclides include tritium and carbon-14.

In the marine environment, the dominant pathway will involve ingestion of seafoods. Fish species tend to be relatively low and only rarely greater than  $10 \text{ Bq} \cdot \text{kg}^{-1}$  polonium-210. Crustacean species tend to have polonium-210 concentrations of  $10\text{-}50 \text{ Bq} \cdot \text{kg}^{-1}$ . Mussels and winkles have similar concentrations. An assessment of dose from all naturally-occurring radionuclides made in Project Marina (Commission of the European Communities [CEC] 1990) indicated that individuals who have high sea-food consumption

rates (e.g., fish  $600 \text{ g} \cdot \text{d}^{-1}$ ; crustaceans, mollusks and seaweed  $100 \text{ g} \cdot \text{d}^{-1}$ ) would receive an annual dose of about 2 mSv. The overwhelming contributor to this dose would be polonium-210 from the molluscan part of the diet.

## Radiation Dosimetry

### Dose to Man

In order to limit radiation exposure of the general public on the basis of the constraints and conditions recommended by the International Commission on Radiological Protection (ICRP), and to limit radiation exposure to marine organisms, it is necessary to develop methods for relating the magnitude of the potential radiation source to the resultant dose. Because of the uncontrolled nature of the interactions involved in the transport of radionuclides from the source into components of the environment, there are only two points at which control can be applied for disposing of liquid radioactive wastes into coastal waters or packaged waste into deep oceans: at the point of release and at the point of exposure.

Dose can rarely be determined directly. In order to apply control at the point of release, it is necessary to first establish the relationship between doses, concentration in environmental materials, and release rates by modelling. To achieve this, a mathematical model (or set of models) is formulated from the available data or, if data are lacking, from realistically conservative (restrictive) estimates. The model must account for the physical transport, geochemical cycling, and ecological transfer of the radionuclides in order to determine, for human exposure ingestion, inhalation and external exposure (Templeton and Preston 1982, IAEA 1983).

The calculations of external, inhalation and ingestion dose are relatively simple once the concentrations in water, sediment and biological materials have been established and occupancy rates for the individuals at potential risk determined. The basic dosimetric models and parameters for human exposure have been developed by ICRP.

### Intervention

ICRP Publication 60 (ICRP 1991) contains the latest recommendations of the Commission. These deal with practices that cause or increase the exposure of individuals to ionizing radiation and with interventions which reduce such

exposure. Where the occurrence of exposures is foreseen, control can be applied at the source to limit the exposure. However, when high-level nuclear fuel and packaged wastes are dumped into the sea in a manner contrary to international requirements, control procedures would be difficult to apply. In this case, when a dose assessment prediction indicates that there may be exposures in the future that approach those that would cause severe effects, countermeasures may be called for. Programs of intervention need to be justified to demonstrate that they do more good than harm. Their form, scale and duration should be optimized to maximize the net benefit (ICRP 1993).

#### Dose to Organisms

Much attention has been given to the process required to limit the radiation exposure of the general public as recommended by ICRP. However, similar constraints have not been applied in the past to protect the environment. In this regard the National Council on Radiation Protection and Measurements (NCRP 1991) and the IAEA (1992) have suggested that the radiation dose to organisms should not exceed  $1\text{--}10\text{ mGy}\cdot\text{d}^{-1}$  ( $0.1\text{--}1.0\text{ rad}\cdot\text{d}^{-1}$ ). They suggest that this value is appropriate because the concern is with populations of organisms, rather than with individuals, as it is with man.

An assessment (IAEA 1988) of the impact of deep-sea dumping of low-level waste on living marine resources based upon the IAEA definition indicated that mollusks living on the sea bed in the dumping area may receive about  $0.1\text{ mGy}\cdot\text{d}^{-1}$ , or a dose rate which is about 2.5 times that of the background upper bound. This dose rate would result in no discernable environmental damage (Nuclear Energy Agency [NEA] 1985).

#### Dumping at the North East Atlantic Dump Site

Between 1948 and 1982, eight European countries conducted radioactive waste dumping operations in the Northeast Atlantic Ocean. No dumping has been conducted since 1982 in accordance with the moratorium agreed to at the London Convention 1972. The operations were conducted subject to regulations by the appropriate national authorities and within the guidelines and recommendations specified by the London Convention 1972, the NEA of the Organization of Economic and Cooperative Development (OECD), and the IAEA. Reviews of the site suitability were conducted by NEA every five years. The NEA published

such a review and dose assessment (NEA 1985), as required by the *P & Multilateral Consultation and Surveillance Mechanism for Sea Dumping of Radioactive Wastes* (OECD 1977). The sources of radioactive waste were low-level wastes from nuclear power operations, other nuclear fuel cycle operations including reprocessing, radionuclides used in research, medicine and industry, and wastes arising from decommissioning of redundant plants and facilities. The sites utilized over the years were all below 50°N and in depths exceeding 4000 meters. The quantities of radioactive waste were about 666 TBq of alpha-emitters; 26146 TBq of beta/gamma emitters and 15474 TBq of tritium. In comparison, the United States dumped about 3.5 PBq (95 KCi) in the Atlantic and Pacific Oceans.

The radiological assessment (Templeton 1981) conducted by the multinational Coordinated Research and Surveillance Program (CRESP) for NEA followed the road-map given in Fig 1. A source term model estimating the release rate was developed because surveillance data indicated no significant concentrations in water, biota and sediment. Because of the sites' remote position from and the long time scale involved in deep-ocean dispersion models, an oceanographic model of the world's oceans was developed (Camplin and Hill 1986). The model predicted radionuclide concentrations in water and sediment as a function of time, and this data was then used to estimate individual doses to members of a critical group. The peak annual doses for past dumping practices via actual pathways were about  $2 \times 10^{-5}$  mSv. The dominant contributors to those doses were plutonium-239 and americium-241. For both the radionuclides, mollusc consumption in the Antarctic by a hypothetical population was the major exposure pathway with the peak dose reached at times between 100 and 500 years after the start of dumping operations. Individual doses were also calculated for the consumption of deep-water fish from the Northeast Atlantic although no such fishery exist today. The highest individual annual dose rate from consumption of these fish was estimated to be  $2 \times 10^{-4}$  mSv at 50 years, and the major contributor was plutonium-239. The appropriate ICRP dose limit for members of the public is  $1 \text{ mSv a}^{-1}$ .

Models used to calculate concentrations of radionuclides in the ocean are described in Appendices VI and VII of the GESAMP Report (IAEA 1983) and summarized in the IAEA report entitled *The Oceanographic and Radiological*

Parts of System Included in Model	Model	Major Processes Included in Model
Canister and lining Waste form	<div style="border: 1px solid black; padding: 5px; margin-bottom: 10px; text-align: center;">Waste Package</div> <p style="text-align: center;">↓ rates of release of radionuclides into the ocean, as a function of time</p> <div style="border: 1px solid black; padding: 5px; margin-bottom: 10px; text-align: center;">Ocean Dispersion and Sedimentation</div> <p style="text-align: center;">↓ radionuclide concentrations in water and sediments, as a function of time</p> <div style="border: 1px solid black; padding: 5px; margin-bottom: 10px; text-align: center;">Dose to Man and Organisms</div>	Canister corrosion Degradation of package lining and caps Release of radionuclides from waste forms
Bottom sediments Benthic boundary layer (water and particulates) Open ocean (water and suspended particulates) Coastal waters		Diffusion and advection  Interactions between radionuclides and suspended particulates and bottom sediments
Exposure pathways - seafoods, beaches, atmosphere, salt, water Dose to marine organisms		Reconcentration of radionuclides in marine organisms, beach sediments, and atmospheric aerosols/vapour Radionuclide intake and metabolism by man and organisms

Figure 1. Modelling Framework Used in Radiological Assessment (after OECD/NEA 1985b)



*Basis for the Definition of High Level Wastes Unsuitable for Dumping at Sea (IAEA 1984).*

Dumping and Disposal in the Barents and Kara Seas

In 1991, information about the Soviet Union's practice of dumping radioactive waste in the Arctic Seas became available from the international organization *Greenpeace*. This material was presented to the London Convention 1972 so that the IAEA could correct its inventory of radioactivity dumped into the world's seas as required by the London Convention 1972. In 1992 the President of the Russian Federation convened a Commission on Matters Related to Radioactive Waste Disposal at Sea. This Commission reported to the President in 1993 (Yablokov et al, 1993). The Commission's report suggests that the total amount of solid radioactive waste dumped into the Northern Arctic Seas was about 903 TBq (24 kCi) of which 0.0007 TBq (0.2 Ci) was dumped into the Baltic; 3.7 TBq (100 Ci) in the White Sea; 450 TBq (12 kCi) into the Barents Sea; and 315 TBq (8500 Ci) into the Kara Sea. The composition of these wastes included small ships, barges, packaged wastes and redundant equipment. Liquid radioactive wastes discharged into the Barents and Kara Seas and the Ara Bay were about 651 TBq (17600 Ci). The quantity of spent nuclear fuel (predominantly fission products and actinides) expressed as strontium-90 equivalents, was about 1704 TBq (4600 kCi). This material was dumped in protected packages and in nuclear submarines.

The IAEA was requested by the Contracting Parties to the London Convention 1972 to pursue a program to assess the risks to human health and the environment. The IAEA was to examine possible remedial actions related to the dumped wastes and to advise on whether they were necessary and justified. In 1993 the IAEA, with the Norwegian Radiation Protection Authority and the Scientific Production Association-TYPHOON of the Russian Federation, organized an international meeting to initiate this program. The main objectives of the meeting were 1) to review available information relevant to the dumping of radioactive wastes into the Barents and Kara Seas and 2) to launch an international four-year program entitled the International Arctic Seas Assessment Project (IASAP) for assessing the existing and possible future radiological and environmental impacts of the dumping and for examining whether remedial actions are necessary. Information in the Yablokov Report

(1993) and the results of the joint Norwegian-Russian Federation cruise to the Barents and Kara Seas in 1992 were not available at that time. Dose assessments conducted by the Russian Federation for man and organisms were presented. It was assumed that the radioactivity, and hence doses, received from consumption of fish from the Barents Sea originated from Sellafield (UK) and Cap de la Hague (France).

As a result of congressional action, the United States has also initiated a comparable assessment program through the U.S. Navy Office of Naval Research. This program will likely be coordinated with IAEA towards achieving the objectives of IASAP.

### Research Needs

A dose assessment of the present and potential impact of the disposal and dumping operations conducted in the Barents and Kara Seas is extremely complex. Not only are we dealing with high-level radioactive wastes as nuclear fuel in reactors but also with activation products in the dumped structures, other packaged waste of indeterminate composition, and liquid wastes from naval operations. This material has been dumped in relatively shallow waters on the shelf surrounding the island of Novaya Zemlya. While the surveillance data today does not suggest a significant contribution from these sources, the potential for releases in the future needs to be examined in great detail. First, we must establish whether leaving the high-level wastes in their present placement presents any risk to man or marine organisms. Secondly, if a risk has been demonstrated, we must determine what intervention will be necessary to reduce that risk. To complicate matters, we not only have an identified contribution in the Barents Sea from the radionuclides discharged from chemical reprocessing plants at Sellafield (UK) and Cap de la Hague (France), but we have contributions to the Kara Sea from actual and potential releases from the military weapons complexes in the watersheds (~5 million km<sup>2</sup>) of the Ob and Yenesi Rivers. Research needs are shown below by category.

### Source Terms and Release Rates

- The initial needs are for detailed information on the radionuclide composition of the reactor fuel in the dumped reactors and the

composition of the activation products in the dumped submarines and in the packaged wastes. In order to develop release rates from these modules, including the furfural matrix around the reactor cores, corrosion rates under Arctic conditions are essential for the variety of materials.

- Information will be required on past and present quantities and composition of the input from the major rivers entering the Barents and Kara Seas. Because of the potential for increased releases in the future from the reactor and from reprocessing and waste disposal sites in these watersheds, some predictive modelling will be required.
- The objective of this task is to develop 1) an inventory of the dumped materials, 2) a realistic release rate model for the packaged dumped waste including that for the reactor fuel, and 3) a screening model for the potential contributions from the Kara and Barents Seas watersheds.

#### Concentration and Distribution of Radionuclides in the Arctic

The objective of the task is to:

- Assemble and analyze all data on concentrations of radionuclides, water, sediment and biota from the Arctic Ocean.
- Predict the present and future contribution to the Arctic Seas from the low-level liquid effluents from European reprocessing plants.

#### Transport of Radionuclides in the Arctic Oceans

- The present state of knowledge on the oceanography of the area needs to be reviewed. Some simple models have been developed, but data needs should be assessed and composite models developed. These models should include the geochemical interaction between radionuclides and suspended sediments and bottom sediments.

- The role of ice scour on sediments and packaged wastes and ice pack development on dispersion needs to be considered. This determination is particularly important in the Kara Sea.
- The objective of this task must be to provide a definitive model to provide output on the concentrations of radionuclides in seawater and sediments, both spatially and temporally.

#### Ecological Characteristics and Bioaccumulation

- The Arctic marine ecosystem is clearly very different from that found in the more temperate Atlantic and Pacific Oceans (Fig 2). Differences in biological productivity, energy transfer, metabolic physiologic rates, and rates of bioaccumulation present a challenge to the radioecological community. We have little evidence, for instance, that the bioaccumulation factors presently used, can be relied upon in an Arctic assessment. There is an essential need to review the existing data and develop appropriate sampling strategies to obtain reliable parameters for use in this assessment. Some radionuclides of interest in this assessment are already present in the Arctic environment from weapons test fallout or from the European discharges from reprocessing plants. A limited surveillance program may be useful to determine the distributions of the radionuclides and their interrelationships with sediment and biota to determine whether they are markedly different from those found in more temperate regions.
- The objective of this task is to provide, based upon the output from the Oceanographic Model, a data base on the predicted concentrations of radionuclides in all significant Arctic food-chain pathways.

#### Pathways Analyses

The design of effective and economical assessments depends heavily on identifying those critical groups which are representative of the individuals expected to receive the highest doses, and those critical radionuclides and exposure pathways which are responsible for most of the dose received by critical groups (Templeton 1981). Furthermore, information will be required

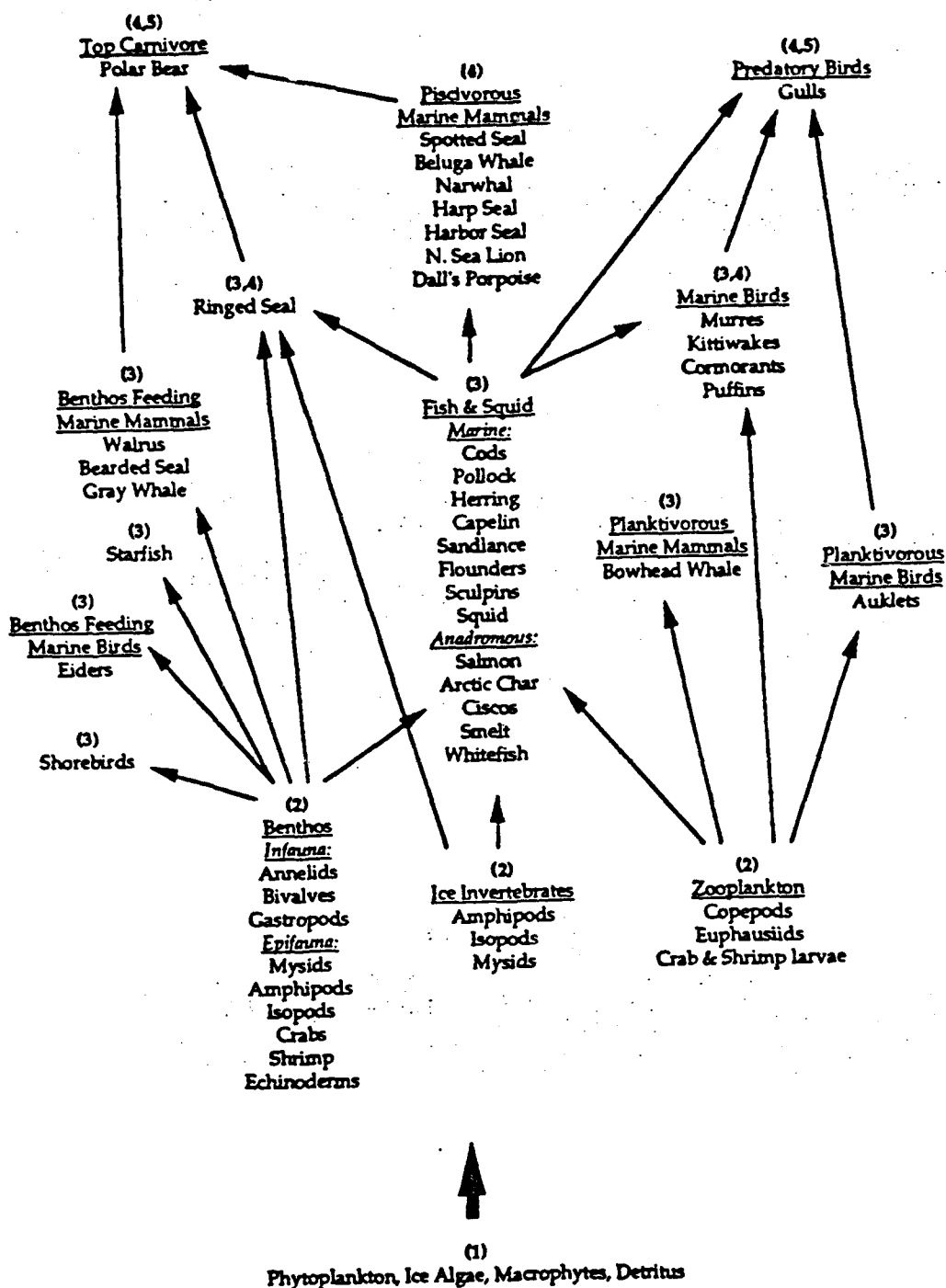


Fig. 2. Generalized composite food web of the Arctic marine ecosystem. The number in parenthesis indicates trophic level in ascending order. Examples of each major category of biota are also listed. (Paul Becker. 1993. IAPRC. Anchorage, AL)

on the pathways responsible for significant collective dose commitments. These may not necessarily be the same as those which are important with respect to the exposure of individuals, and their additional pathways will have to be taken into account. When applying ICRP or national dose limits, it is necessary to ensure beyond a reasonable doubt that the dose to the most exposed individuals is within the dose limits. Therefore, the working, eating and recreational habits of the local populations and populations at some distance from the site must be identified. These studies should include items listed below.

- The type and amounts of ingested marine sea-food derived from specific areas should be estimated. As can be seen from Fig 2, intensive radioecological studies will be necessary to elucidate food chain relationships and seasonal food consumption by humans, over a wide selection of mammals, fish and invertebrates.
- The majority of the indigenous peoples of the Arctic Rim depend heavily upon the arctic tundra environment for food. Because these terrestrial animals and plants have been shown to have elevated levels of fallout and natural radionuclides, their contribution needs to be factored into the dose assessment.
- For external dose calculations, it is necessary to estimate the number of hours spent handling fishing gear at sea and on the beach, and the number of hours spent on the beach and on the ice at work and for recreation.
- Consumption and occupancy rates derived from habit surveys of an identified critical group are not normally distributed. These variations can be accounted for by applying the appropriate dose limit for individual members of the public to the weighted mean dose equivalent for the group.
- A dose integration and assessment group should be established early in the assessment project. This group should lay out the project road-map.

Experience has shown that this group needs to identify the objectives of the assessment and provide guidelines to the source term, oceanographic, geochemical, ecological and human factors groups on the scope and type of information required. Without this input, the input data to the integration and assessment group may be inadequate and/or inappropriate. This group should initiate screening studies to assist in defining the major radionuclides and pathways.

- Studies on alternative countermeasures to reduce the potential for radiation exposures from radioactivity dumped and disposed of in the Barents and Kara Seas should be initiated. With this in hand, a strategy for intervention should be developed.

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**RADIONUCLIDE CONCENTRATIONS IN BIVALVES COLLECTED  
ALONG THE COASTAL UNITED STATES**

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**ABSTRACT**

In 1990, the National Oceanic and Atmospheric Administration's National Status and Trends Program initiated a study of artificial radionuclides ( $^{241}\text{Am}$ ,  $^{239+240}\text{Pu}$ ,  $^{238}\text{Pu}$ ,  $^{137}\text{Cs}$ ,  $^{90}\text{Sr}$ ,  $^{110}\text{Ag}$ ,  $^{65}\text{Zn}$ ,  $^{60}\text{Co}$ , and  $^{58}\text{Co}$ ) in oysters and mussels collected along the coastal United States. The results show that  $^{241}\text{Am}$  and  $^{137}\text{Cs}$  concentrations are the highest along the West Coast of the United States whereas  $^{238}\text{Pu}$ ,  $^{239+240}\text{Pu}$ , and  $^{90}\text{Sr}$  appear to be more concentrated in bivalves collected on the East Coast. Activation products,  $^{110}\text{Ag}$ ,  $^{65}\text{Zn}$ ,  $^{60}\text{Co}$ , and  $^{58}\text{Co}$ , were sometimes present close to nuclear facilities. When our results for  $^{239+240}\text{Pu}$  and  $^{137}\text{Cs}$  are compared to those of an earlier (1976 through 1978) Mussel Watch Program sponsored by the Environmental Protection Agency, they show a statistically significant decrease in the concentrations between the mid 70s and the early 90s ( $P < 0.05$ ). For  $^{241}\text{Am}$  no statistical differences were detected.

## INTRODUCTION

The National Oceanic and Atmospheric Administration's (NOAA's) National Status and Trends Program (NS&T), was initiated in 1984 to monitor the environmental quality of United States (US) coastal and estuarine areas. The program includes two main monitoring projects: the National Benthic Surveillance Project, initiated in 1984, that collects and analyzes sediments and bottom fishes from 149 sites and the Mussel Watch Project (MWP), initiated in 1986, that collects and analyzes bivalves (mussels and oysters) and sediments from over 250 sites. Collected samples are analyzed, on a yearly basis, for 14 metals (major and trace elements) and 70 organic contaminants (polyaromatic hydrocarbons, polychlorinated biphenyl congeners, pesticides, and butyltins).

Because bivalves concentrate contaminants while filtering their surrounding waters, these organisms are useful indicators of changes occurring in the chemistry of their environment. Their response to chemical changes in their surroundings is noticeable within a matter of days with approximately 4 to 24 weeks required for the bivalves to equilibrate with their environment (Roesijadi *et al.*, 1984).

In 1990, bivalve samples were collected at 36 sites (Fig. 1) and analyzed for the radionuclides:  $^{241}\text{Am}$ ,  $^{239+240}\text{Pu}$ ,  $^{238}\text{Pu}$ ,  $^{137}\text{Cs}$ ,  $^{110}\text{Ag}$ ,  $^{90}\text{Sr}$ ,  $^{65}\text{Zn}$ ,  $^{60}\text{Co}$ ,  $^{58}\text{Co}$ ,  $^{40}\text{K}$ , and  $^7\text{Be}$ . Most of these 36 samples were obtained from MWP sites and a few additional samples were collected in the vicinity of nuclear facilities or known radioactive dumping sites. Three molluscan species were collected for this project: the complex species *Mytilus edulis*, along the North Atlantic and Pacific coasts, *Mytilus californianus*, generally found in high energy environments from the Pacific Coast, and *Crassostrea virginica*, collected along the Atlantic shore from Delaware Bay to Florida and along the Gulf of Mexico coast (Table 1).

Between 1976 and 1978, the Environmental Protection Agency (EPA), conducted a Mussel Watch Program (MWP70s) that measured transuranic elements, polyaromatic hydrocarbons, polychlorinated biphenyls, chlorinated pesticides and trace metals in bivalves collected around the country (Farrington, 1983; Farrington *et al.*, 1983; Goldberg *et al.*, 1978, 1983; Palmieri *et al.*, 1984). It is possible to compare the data obtained in the late 70s and in the 90s at 30 of the 36 sites and for 4 radioisotopes i.e.,  $^{241}\text{Am}$ ,  $^{239+240}\text{Pu}$ ,  $^{238}\text{Pu}$ , and  $^{137}\text{Cs}$ .

Table 1 NS&amp;T Site Locations

SITE NO.	MAIN LOCATION	SPECIFIC LOCATION	STATE	SPECIES <sup>a</sup>
	1*	Merriconeag Sound	ME	ME
	2*	Boston Harbor	MA	ME
E	3*	Duxbury Bay	MA	ME
A	4*	Block Island Sound	RI	ME
S	5*	Long Island Sound	NY	ME
T	6*	Hudson/Raritan Estuary	NY	ME
	7*	Absecon Inlet	NJ	ME
	9	Delaware Bay	DE	CV
C	8*	Chesapeake Bay	VA	CV
O	10	Chesapeake Bay	MD	CV
A	11*	Cape Fear	NC	CV
S	12*	Savannah Riv. Estuary	GA	CV
T	13	Indian River	FL	CV
	14	Biscayne Bay	FI	CV
	15*	Cedar Key	FL	CV
G	16*	Apalachicola Bay	FL	CV
U	17*	Barataria Bay	LA	CV
L	18*	Galveston Bay	TX	CV
F	19*	Matagorda Bay	TX	CV
	20*	Oceanside	CA	ME
	21*	La Jolla	CA	MC
W	22	Newport Beach	CA	MC
E	23*	San Pedro Harbor	CA	ME
S	24	Santa Cruz Island	CA	MC
T	25*	San Luis Obispo Bay	CA	MC
	26*	Pacific Grove	CA	MC
	27*	San Francisco Bay	CA	ME
	28*	San Francisco Bay	CA	ME
	29*	Farallon Islands	CA	MC
C	30*	Bodega Bay	CA	MC
O	31*	Humboldt Bay	CA	MC
A	32*	Crescent City	CA	MC
S	33*	Yaquina Bay	OR	MC
T	34*	Columbia River	OR	ME
	35*	Grays Harbor	WA	MC
	36*	Whidbey Island	WA	ME

<sup>a</sup>ME - *Mytilus edulis*, MC - *Mytilus californianus*, CV - *Crassostrea virginica*

\*Common sites to NOAA NS&T and to EPA Mussel Watch Project (MWP70s), see here after.

The first test of a nuclear weapon occurred in July 1945 at Alamogordo, New Mexico. Following Nagasaki and Hiroshima (August 1945), several US nuclear devices were detonated in islands of the Pacific Ocean or at the Nevada Test Site (Carter and Moghissi, 1977). In the early 1950s, there was an active period of atmospheric weapon testing by the US, United Kingdom (UK), and the Soviet Union (USSR). A moratorium on nuclear testing was initiated in November 1958 and ended in September 1961. The pre-moratorium period was dominated by US weapon testing and the post-moratorium period was dominated by USSR weapon testing. From 1966 to 1974 most of the atmospheric tests were conducted by France in the Tuamotu Archipelago (Mururoa and Fantataufa Islands) and by the People's Republic of China (Koide *et al.*, 1979, 1985). In 1974, France joined the US and USSR in conducting only underground tests and, in 1980, the People's Republic of China is believed to have followed suite. India detonated its first weapon underground in 1974. Presently, all the known nuclear testing is underground, including the large event recorded in May 1992 from the People's Republic of China (NRDC, 1991-P. Davis, Personal Communication).

The phase of intense atmospheric nuclear testing resulted in the injection of fission and fusion products into the upper atmosphere and in a global fallout of radionuclides including  $^{241}\text{Pu}$  ( $T_{1/2} = 13$  years) and its daughter product  $^{241}\text{Am}$ ,  $^{239+240}\text{Pu}$ ,  $^{238}\text{Pu}$ ,  $^{137}\text{Cs}$ , and  $^{90}\text{Sr}$ .

In 1964, there was an additional input of  $^{238}\text{Pu}$  to the atmosphere due to the burn-up of the SNAP-9A satellite that was fueled by plutonium (Mamuro and Matsunami, 1969; Koide *et al.*, 1977).

The occurrence of the radioisotopes  $^{241}\text{Am}$  ( $T_{1/2} = 458$  years),  $^{239+240}\text{Pu}$  ( $T_{1/2} = 24400$  years and 6580 years respectively),  $^{238}\text{Pu}$  ( $T_{1/2} = 86$  years),  $^{137}\text{Cs}$  ( $T_{1/2} = 30$  years), and  $^{90}\text{Sr}$  ( $T_{1/2} = 28$  years) in the environment is mostly derived from global fallout associated with this intense period of atmospheric testing and from remobilization of soil particles. Nuclear reactor accidents, such as Chelyabinsk (September 1957) and Chernobyl (April 1986) in the USSR, have also released isotopes such as  $^{137}\text{Cs}$  to the atmosphere. However, because these inputs were released into the lower atmosphere, their impact appears to be primarily limited to nearby areas (few hundreds of Km- Mélière *et al.*, 1988; Pyatt and Beaumont, 1992).

Some artificial radionuclides such as  $^{137}\text{Cs}$  and  $^{90}\text{Sr}$ , and also  $^{110}\text{Ag}$  ( $T_{1/2} = 253$  days),  $^{65}\text{Zn}$  ( $T_{1/2} = 243.6$  days),  $^{60}\text{Co}$  ( $T_{1/2} = 5.3$  years), and  $^{58}\text{Co}$

( $T_{1/2} = 71.3$  days), are formed by activation in nuclear reactors and can be released to the environment by nuclear power plant cooling waters.

Potassium - 40 ( $T_{1/2} = 1.28 \times 10^9$  yrs) occurs naturally and  $^7\text{Be}$  ( $T_{1/2} = 53.4$  days) is formed in the upper atmosphere by cosmic ray bombardment of oxygen and nitrogen nuclei. The results obtained for the  $^{40}\text{K}$  analysis will mostly be used as reference.

The aim of this study is to determine the present status of radionuclide contamination in the coastal and estuarine environments of the United States and to document the changes that have occurred in this contamination over the last 15 years.

## **ANALYTICAL TECHNIQUES**

In order to obtain about 300 g of dry soft tissue, approximately 180 to 200 mussels or at least 125 to 150 oysters were collected at each site. Unfortunately, in some cases, due to the small size of the mollusks, less than 100 g of dried sample were recovered. After collection, the animals were packed in plastic containers and frozen on dry ice until shucked for analysis. The samples were shipped overnight to Texas A&M University, Geochemical and Environmental Research Group (GERG) where they were shucked and the soft tissues freeze dried and weighted. The dried samples were then shipped in sealed glass containers to Thermo Analytical Inc., California Laboratories (TMA/Norcal).

At TMA/Norcal, the samples were redried, reweighed, charred and ashed at  $425^\circ\text{C}$ .

After the ashing step was completed, the entire sample was placed into either a 100 mL jar or a 15 mL petri dish for counting. The geometries were double checked to verify accurate and current calibrations. Counting was performed with high resolution, low background gamma spectrometers. High purity germanium (HPGe) detectors were used to analyze for  $^7\text{Be}$ ,  $^{40}\text{K}$ ,  $^{58}\text{Co}$ ,  $^{60}\text{Co}$ ,  $^{65}\text{Zn}$ ,  $^{110}\text{Ag}$ , and  $^{137}\text{Cs}$ . The typical background for  $^{60}\text{Co}$  was about 0.08 cpm and for  $^{137}\text{Cs}$  of about 0.07 cpm. A standard Nuclear Data peak search routine (ND6620) was used to identify gamma emissions, and calculations were done using a TMA/Norcal program and double checked by the staff. This included: verifying data inputs (e.g., aliquot, reference time, geometry, etc),

reviewing the automated peak search routine, verifying peak subtraction, and double-checking the peak library to verify that all the isotopes present in the sample were accounted for.

After  $\gamma$  counting, the ashed material was dissolved and  $^{241}\text{Am}$ ,  $^{239+240}\text{Pu}$ ,  $^{238}\text{Pu}$ ,  $^{137}\text{Cs}$ , and  $^{90}\text{Sr}$ , were chemically separated. Techniques used to quantify the radionuclides of interest are discussed in Wessman *et al.* (1971, 1977, 1978). Because of the high sensitivity needed for the analysis, sequential determination was performed. A brief discussion of the methods used for chemical separations and counting follows (after TMA/Norcal 1991, 1992).

After an initial leaching with nitric acid ( $\text{HNO}_3$ ) and hydrogen peroxide, the samples were filtered and the filters were ashed and dissolved using a mixture of concentrated hydrofluoric acid ( $\text{HF}$ ) +  $\text{HNO}_3$  + hydrochloric acid ( $\text{HCl}$ ). All dissolution fractions were combined, dried, and dissolved in 8N  $\text{HNO}_3$ .

Aliquots (80% of the total solution) were taken,  $^{242}\text{Pu}$  and  $^{243}\text{Am}$  tracers and yttrium carrier were added, the solutions were equilibrated and the volume of the 8N  $\text{HNO}_3$  solution reduced.

Plutonium was extracted initially from the sample in nitric solution on a large-scale AG 1 x 8 anion resin column and eluted with  $\text{HNO}_3$  +  $\text{HF}$ . The eluant was then further purified on a second small scale nitrate column of AG 1 x 8 anion resin. The purified fraction was electrodeposited onto a 1" stainless steel disc and submitted for alpha spectrometry. Each alpha spectrum was obtained from one of the 26 solid state 450  $\text{mm}^2$  surface barrier diodes used by TMA/Norcal. Each detector utilizes 256 channels in a Nuclear Data ND66 computer controlled multichannel analyzer system. The sample was counted for at least 1000 minutes with the spectra collected in 256 channels over the 3.6 to 7.00 MeV energy region. Energy calibration sources were counted before and after the sample to set peak integration limits. A background measurement and evaluation program was maintained for each detector, with backgrounds ranging usually from 0.0005 to 0.004 cpm within the regions integrated for the spectrometric analysis. The efficiencies varied from 24 to 33%.



**Table 2 - MDA and Precision Achieved for the Radioisotopes Measured In This Project ( $\times 10^{-3}$  pCi).**

Type of analysis	MDA*	Precision (+/- %)**		
$\alpha$ emission		0-100**	100-300**	>300**
$^{241}\text{Am}$	0.081 to 2.6	14	12	8
$^{239-240}\text{Pu}$	0.980 to 2.5	24	7	4
$^{238}\text{Pu}$	0.068 to 6.2	2	27	3
$\beta$ emission				
$^{137}\text{Cs}$	3.0 to 11	4	11	4
$^{90}\text{Sr}$	1.3 to 34	6	17	13
$\gamma$ emission				
$^{137}\text{Cs}$	11 to 210	1 (87)***		
$^{110}\text{Ag}$	26 to 470	1 (15)		
$^{65}\text{Zn}$	47 to 76	2 (37, 67)		
$^{60}\text{Co}$	16 to 260	1 (98)		
$^{58}\text{Co}$	11 to 140	1 (12)		

\*From the ANSI N13.30 bioassay standard, the MDA is defined as: "The smallest amount of analyte radionuclide in a sample that will be detected with alpha beta probability of non-detection (Type II error) while accepting an alpha probability of erroneously detecting that radionuclide in an appropriate blank sample (Type I error).

$$\text{MDA} = \frac{4.65 \sqrt{\text{BKG counts}} + 3}{D \times Y \times E \times W \times 2.22}$$

4.65 = Constant, BKG = Background counts, 3 = Constant,  
2.22 = Conversion from dpm to pCi, E = Detector efficiency,  
W = Sample dry weight, T = counting time, D = decay correction  
from counting time to time of collection, Y = yield of the  
chemical recovery

\*\*Precision categories for the analyses presented in this study: the numbers in the columns represent the number of analyses with precision falling into that category.

\*\*\*Precision of the analysis for which the isotope activity was above detection limits.

Strontium-90 was analyzed by isolating the  $^{90}\text{Y}$  daughter product ( $T_{1/2} = 64$  hours) and beta counting up to five times over a period of two weeks. A least squares regression was used to calculate  $^{90}\text{Y}$  at separation and equilibrium was assumed in the calculation of  $^{90}\text{Sr}$  activity. Chemical purification was performed using DDGP (n,n-diethyl dicarbamoyl phosphonate) extraction out of the column load fraction from the initial Pu column. The yttrium was back-extracted into dilute  $\text{HNO}_3$ . Hydroxide, fluoride, and oxalate precipitations were performed for further purification. A mixed nitric acid + alcohol anion column was run to separate the Am species which had been carried with the yttrium to

this point. Further precipitations were performed and the purified yttrium fraction was mounted on a planchet as an yttrium oxide ( $\text{Y}_2\text{O}_3$ ), weighed for chemical recovery and beta counted on a gas flow proportional detection system. The yttrium planchets mounted for  $^{90}\text{Sr}$  analysis had default counting instructions for a 200 minute first count. In actuality, the great majority had a 400 minute first count. This first count was used in determining the detection limit (Table 2). One hundred to 200 minute counts were repeated until the count rate dropped below 0.2 counts per minute (cpm) or the count rate became indistinguishable from background. The interval of counting varied from one to three days. Most samples received only two or three total beta counts since the activities of these samples were low. The beta counting is described further under the cesium section.

Because the  $\gamma$ -counting of the samples for  $^{137}\text{Cs}$  generally gave values close to or below detection limits (Table 2),  $^{137}\text{Cs}$  was analyzed by  $\beta$ -counting after radiochemical purification in the 19 samples for which there was enough material left. In these cases, the remaining sample aliquot (20% of the original sample) was taken, cesium carrier added, and the solution adjusted to a pH of 1.0. The cesium fraction was carried on ammonium molybdophosphate crystals, dissolved in a basic EDTA solution and run through a Biorex-40 cation exchange column. The dilute HCl eluant was concentrated and further purified by precipitation of cesium silicotungstate and, finally, cesium chloroplatinate. This precipitate was mounted, weighed for recovery, and submitted for beta counting. Each  $^{137}\text{Cs}$  measurement was on one of the 14 low background Geiger-Muller detectors on-line. The backgrounds were approximately 0.5 cpm and the data from each detector were corrected for individual differences in detector sensitivity. The system was directly connected to the PDP 11/84 computer for data acquisition and transfer. The nominal efficiency for  $^{137}\text{Cs}$ , including self absorption effects for precipitate thickness, was approximately 35%.

A very strict quality assurance program (QAP) was applied during these analyses. In particular, all the chemical separations were performed in a "low level laboratory" which is restricted to samples with activity ranging from zero to few pico-curies (pCi). In addition, environmental and "zero level" samples were treated in dedicated modules inside the low level laboratory. Special coats, equipment, material and reagents were reserved to this area. In particular, brand-new glassware and Teflon containers were used for this project. Cross

contamination was prevented by segregating the samples by activity level upon their arrival in the laboratory. The "Quality Assurance Manual for TMA/Norcal Programs" details the TMA/Norcal QAP that includes: personnel training and proficiency, instrument background and stability checks, blanks, standards, calibrations, procedure approval, data verification, etc.

"Laboratory Intercomparison Studies Program" for various matrices and including  $\alpha$ ,  $\beta$  and  $\gamma$  emitters isotopes were routinely performed. Standards from the National Institute of Standards and Technology (NIST), the United Kingdom Atomic Energy Authority (UKAEA) and others were processed on a routine basis. Finally TMA/Norcal had participated in several collaborative programs of procedure testing and standardization of reference material.

## RESULTS

The data obtained by NOAA NS&T are displayed in Fig. 2 (a, b, c, d) and Fig. 3 (a, b, c, d). Figures 2 and 3 show that, along the US coasts, the fallout radioisotope concentrations range from:  $1 \times 10^{-5}$  to  $243 \times 10^{-5}$  pCi/g for  $^{241}\text{Am}$  (Fig. 2a), from  $7 \times 10^{-5}$  pCi/g to  $239 \times 10^{-5}$  pCi/g for  $^{239+240}\text{Pu}$  (Fig. 2b), from  $0.1 \times 10^{-5}$  to  $160 \times 10^{-5}$  pCi/g for  $^{238}\text{Pu}$  (Fig. 2c), from  $46 \times 10^{-5}$  to  $1080 \times 10^{-5}$  pCi/g for  $^{137}\text{Cs}$  (Fig. 2d), and from  $70 \times 10^{-5}$  to  $5390 \times 10^{-5}$  pCi/g for  $^{90}\text{Sr}$  (Fig. 3a). All the results are expressed in pCi/g of material (dry weight).

The activation products (Fig. 3b) are generally below detection limits (Table 2) except at the following sites: Chesapeake Bay-Calvert Cliff ( $^{110}\text{Ag} = 0.32$  pCi/g  $\pm 15\%$ ), Delaware Bay-Arnolds Point Shoal ( $^{65}\text{Zn} = 1.01$  pCi/g  $\pm 37\%$ ), La Jolla-Point La Jolla ( $^{65}\text{Zn} = 0.101$  pCi/g  $\pm 75\%$ ), Santa Cruz-Fraser Point ( $^{60}\text{Co} = 0.0273$  pCi/g  $\pm 90\%$ ), and Savannah River-Tybee Island ( $^{58}\text{Co} = 0.250$  pCi/g  $\pm 12\%$ ).

Finally (Fig. 3c, 3d), concentrations of the naturally occurring  $^{40}\text{K}$  range from 5.53 to  $12.40 \times 10^{-5}$  pCi/g and  $^7\text{Be}$  was only detected in one site of the Pacific Northwest (site SGSG).

## INTERPRETATION

### 1) Geographical distribution of the radioactivity in the coastal US

Average and median values have been calculated for the whole set of data (Total) and by coasts (Table 3). Geometric means were also calculated and are generally very close to the median values. For the activation products and  $^7\text{Be}$ , averages and medians were not calculated because only a few of the data points were above the detection limits.

**Table 3 - Distribution of Median, Average, and Standard Deviation of the Radionuclide Activities ( $\times 10^{-5}$  pCi/g, except for  $^{40}\text{K}$  pCi/g).**

	$^{238}\text{Pu}$			$^{239+240}\text{Pu}$			$^{241}\text{Am}$		
	AV*	+/-2 $\sigma$	MED**	AV	+/-2 $\sigma$	MED	AV	+/-2 $\sigma$	MED
<b>TOTAL</b>	16	28	7	42	46	29	61	72	23
<b>EAST</b>	17	16	13	42	42	30	15	8	19
<b>GULF</b>	39	68	13	66	98	21	30	39	16
<b>WEST</b>	8	8	6	33	21	29	100	83	65

	$^{90}\text{Sr}$			$^{137}\text{Cs}$			$^{40}\text{K}$		
	AV	+/-2 $\sigma$ ***	MED	AV	+/-2 $\sigma$	MED	AV	+/-2 $\sigma$	MED
<b>TOTAL</b>	476	968	137	528	297	512	7.94	1.91	7.83
<b>EAST</b>	537	683	234	381	167	399	8.04	2.36	8.16
<b>GULF</b>	339	386	193	173	110	224	8.98	2.53	8.66
<b>WEST</b>	465	1274	134	666	268	740	7.54	1.15	7.81

\* Average value, \*\*Median value, \*\*\*Precision

The West Coast is characterized by higher concentrations of  $^{241}\text{Am}$  and  $^{137}\text{Cs}$  and by lower concentrations of  $^{238}\text{Pu}$  relative to the other US coasts. The average and median values for  $^{90}\text{Sr}$  activities are generally higher in samples collected along the East Coast whereas  $^{238}\text{Pu}$  and  $^{239+240}\text{Pu}$

concentrations are more elevated along the Gulf Coast. As expected for a natural radionuclide,  $^{40}\text{K}$  concentrations show small spatial variation, being moderately elevated along the Gulf Coast and lower along the West Coast.

The short-lived activation products are in general below detection limits (Fig. 3b). Detectable spikes appear only in a few locations: i.e.,  $^{110}\text{Ag}$  in Chesapeake Bay near Calvert Cliffs,  $^{65}\text{Zn}$  in Delaware Bay near Arnolds Point Shoal and in Southern California near Point La Jolla,  $^{58}\text{Co}$  in the Savannah Estuary near Tybee Island, and  $^{60}\text{Co}$  at Fraser Point on Santa Cruz Island. With the exception of Fraser Point, all of these sites are located close to nuclear reactor facilities.

The medians of the  $^{241}\text{Am}$  concentrations are similar for the East and the Gulf coasts ( $19 \times 10^{-5}$  and  $16 \times 10^{-5} \text{ pCi/g}$  respectively) but very different from the medians found for samples collected along the West Coast ( $65 \times 10^{-5} \text{ pCi/g}$ ). Along the Pacific Coast, the values are generally relatively high and variable. The geographic distribution of the concentrations display (Fig. 2a) a maximum located in central California near Point Saint George (CA). In addition to this pattern, isolated large values are evident in California near Point La Jolla ( $189 \times 10^{-5} \text{ pCi/g}$ ), Santa Cruz Island ( $234 \times 10^{-5} \text{ pCi/g}$ ), Monterey Bay ( $183 \times 10^{-5} \text{ pCi/g}$ ), and the Farallon Islands ( $175 \times 10^{-5} \text{ pCi/g}$ ). Along the East Coast, values are almost uniformly low whereas they are more variable along the Gulf Coast. Two of our sites, located near Cedar Key (FL) and Matagorda Bay (TX), show concentrations as high as  $85 \times 10^{-5} \text{ pCi/g}$ .

The median of the concentrations reported for  $^{239+240}\text{Pu}$  is lower for the Gulf ( $21 \times 10^{-5} \text{ pCi/g}$ ) than for the other coasts (about  $30 \times 10^{-5} \text{ pCi/g}$  - Fig 2c). However, along the Gulf of Mexico, the oysters collected near Cedar Key (FL) display one of the highest concentrations ( $239 \times 10^{-5} \text{ pCi/g}$ ) found in our data set. Along the East Coast, several high values are observed. They occur at: Stover Point in Maine ( $127 \times 10^{-5} \text{ pCi/g}$ ), Duxbury Bay in Massachusetts ( $74 \times 10^{-5} \text{ pCi/g}$ ), Absecon Inlet in New Jersey ( $125 \times 10^{-5} \text{ pCi/g}$ ), and Cape Fear in North Carolina ( $77 \times 10^{-5} \text{ pCi/g}$ ). Along the West Coast, the concentrations are low and their geographical distribution displays a maximum in central California. As with  $^{241}\text{Am}$ , the bivalves collected at Santa Cruz Island-Fraser Point (CA), show the highest activities ( $74 \times 10^{-5} \text{ pCi/g}$ ) of  $^{239+240}\text{Pu}$ .

Plutonium-238 concentrations (Fig. 2d) are generally low on all three coasts (median of all the values is  $7 \times 10^{-5} \text{ pCi/g}$ ). Six locations show relatively high concentrations (from 23 up to  $160 \times 10^{-5} \text{ pCi/g}$ ), i.e.: Jamaica Bay (NY),

Savannah Estuary (GA), Biscayne Bay (FL), Cedar Key (FL), Humboldt Bay (CA), and Whidbey Island (WA).

Because analysis after radiochemical separation achieves a lower detection limit, only  $\beta$ -counting results were displayed for  $^{137}\text{Cs}$  (Fig. 2b). In this case, the values obtained for medians and averages calculated using the complete data set (Table 3) were about the same ( $512$  and  $528 \times 10^{-5}$  pCi/g, respectively). In addition, there are two populations of concentrations, one centered around  $36 \times 10^{-5}$  pCi/g and the second around  $93 \times 10^{-5}$  pCi/g. On the West Coast, higher concentrations (up to  $1080 \times 10^{-5}$  pCi/g at Oceanside, CA) are frequently observed.

About 70% of the values reported for  $^{90}\text{Sr}$  are below  $300 \times 10^{-5}$  pCi/g. On the West Coast, the geographic distribution of the concentrations exhibits a less distinct but similar pattern to what has been observed for  $^{241}\text{Am}$  (Fig. 3a). Only one very high value was obtained, near Crescent City-Point Saint George in California ( $5390 \times 10^{-5}$  pCi/g). Along the East Coast, values above  $300 \times 10^{-5}$  pCi/g are more frequent, reaching  $1980 \times 10^{-5}$  pCi/g near Cape Fear (NC). Along the Gulf, the site located in Matagorda Bay (TX) displays the highest concentration ( $1020 \times 10^{-5}$  pCi/g).

From the data presented, it appears that the geographical distribution of the radioisotopes varies from one coast to the other. In particular, the West Coast is characterized by elevated activities of  $^{241}\text{Am}$  and  $^{239+240}\text{Pu}$  in the center part of the coast and Cs may also follow the same pattern. This pattern was previously observed in the EPA MWP70s and at this time it was hypothesized that the enrichment in fallout radionuclides was related to the California Current and to the associated upwelling (Goldberg *et al.*, 1978, 1983; Farrington *et al.*, 1983).

In the ocean, americium, plutonium and cadmium exhibit a surface depletion and an enrichment at intermediate depths (Livingston *et al.*, 1984; Volchok *et al.*, 1971). Upwelling of these intermediate waters to the surface exposes organisms to higher concentrations than are normally found in areas with no upwelling. In the MWP70s study (Goldberg *et al.*, 1983) and in our own studies (O'Connor, 1990, 1992), the high concentrations of cadmium found in the bivalves collected along the central West Coast are supporting the upwelling hypothesis.

## 2) Temporal trends in the radioactivity of the coastal US

When comparing two sets of data obtained by two different laboratories at two different times (about 15 years apart), technical differences that obscure the interpretation of the results can be expected. With this in mind, we have attempted to extract temporal trends from our data by comparing them to the EPA MWP70s. Only  $^{241}\text{Am}$ ,  $^{239+240}\text{Pu}$ ,  $^{238}\text{Pu}$ , and  $^{137}\text{Cs}$  were analyzed consistently in both programs. In general, the MWP70s study consisted of one analysis per year in each of three consecutive years, 1976, 1977, and 1978. Figure 4 (a, b, c, d) compares the means of the EPA MWP70s data with the 1990 NS&T data site by site.

Table 4 compares of the averages and medians calculated for both sets of data. Averages and medians are generally lower in the 90s for  $^{241}\text{Am}$ ,  $^{239+240}\text{Pu}$ , and  $^{137}\text{Cs}$ . The averages and medians are all higher in the 90s for  $^{238}\text{Pu}$ .

In 17 cases out of 28,  $^{241}\text{Am}$  is lower in the 90s study than in the 70s study. Using a non-parametric sign test, no significant difference ( $p < 0.5$ ) can be detected between the results of the two studies. In the case of  $^{239+240}\text{Pu}$ , a decrease has been observed in 20 cases out of 29 and statistically, the probability that a real decrease occurred is greater ( $p < 0.1$ , 90% probability). In all seven cases where comparisons were possible, radiocesium  $^{137}\text{Cs}$  is also significantly lower in the 90s ( $p < 0.01$ , 99% probability).

Taken at face value,  $^{238}\text{Pu}$  results suggest that this isotope may be behaving in a different manner, displaying a significant increase (19 out of 24 cases,  $p < 0.001$ , 99.9 % probability) from the 1970s to the 1990s. At several sites, the 1990 data are almost an order of magnitude greater than in the 70s. However, because these concentrations are very close to the detection limits, the precision is quite variable (the coefficient of variation is often between 100 and 300%), and the accuracy is uncertain. Consequently great prudence must be exercised when using these data and new analyses are necessary to sort out analytical uncertainties from real differences.

**Table 4 - Comparison of the Averages and Medians Calculated for NS&T and EPA MWP70s Data ( $\text{X}10^{-5}\text{pCi/g}$ ).**

	<b><math>^{238}\text{Pu}</math></b>				<b><math>^{241}\text{Am}</math></b>			
	NS&T	+/-2 $\sigma$	EPA*	+/-2s	NS&T	+/-2 $\sigma$	EPA	+/-2s
<b>Aver**. East</b>	17	16	3.5	2	15	8	25	15
<b>Median</b>	13		3.3		19		19	
<b>Aver. Gulf</b>	39	68	5.9	6	30	39	12	5
<b>Median</b>	13		3.6		16		19	
<b>Aver. West</b>	8	8	3.5	2	100	83	170	188
<b>Median</b>	6		2.9		65		96	
<b>Aver. US</b>	16	28	3.8	3	61	72	98	162
<b>Median US</b>	7		3.2		23		34	

	<b><math>^{239+240}\text{Pu}</math></b>				<b><math>^{137}\text{Cs}</math></b>			
	NS&T	2 $\sigma$	EPA	2s	NS&T	2 $\sigma$	EPA	2s
<b>Aver. East</b>	42	42	93	48	381	167	1709	916
<b>Median</b>	30		89		399		1704	
<b>Aver. Gulf</b>	66	98	67	60	173	110	1497	250
<b>Median</b>	21		86		224		1697	
<b>Aver. West</b>	33	21	88	66	666	268	1072	591
<b>Median</b>	29		53		740		1261	
<b>Aver. US</b>	42	48	86	60	528	305	1455	730
<b>Median US</b>	29		74		512		1434	

\*EPA = EPA MWP70s

\*\*Aver. = Average

In the absence of major new sources, the concentrations of most of these isotopes would generally be expected to decrease in the near coastal environment as a combined result of: a) declining input from the stratosphere; b) physical decay; c) burial in shallow depositional sediments; and d) dispersion, dilution and removal to other more remote environmental sinks, such as deep ocean water or sediments. The decrease in input of fallout radionuclides has been documented in undisturbed sediment cores, where the large peak of fallout radioactivity associated with the 1950s and early 1960s declines rapidly in more recently deposited surficial sediments (Noshkin and



Bowen, 1973, 1975; Olsen *et al.*, 1981 a and b; Bopp *et al.*, 1982). Our data for  $^{239+240}\text{Pu}$  and  $^{137}\text{Cs}$  clearly reflect a continuation of this general decreasing trend. The decrease observed for  $^{137}\text{Cs}$  between the mid 1970s and 1990 reaches a factor of 8 (Table 4), compared with a corresponding factor of only about 2 for  $^{239+240}\text{Pu}$ , reflecting the much shorter half-life of  $^{137}\text{Cs}$  (30 years for  $^{137}\text{Cs}$  compared to 6580 years for  $^{240}\text{Pu}$  and 24,400 years for  $^{239}\text{Pu}$ ). The corresponding decline in  $^{241}\text{Am}$  is obscured by the input of new  $^{241}\text{Am}$  from the decay of  $^{241}\text{Pu}$  ( $T_{1/2} = 13$  years), which has been predicted (Livingston *et al.*, 1975, 1976, 1984) to yield  $^{241}\text{Am}/^{239+240}\text{Pu}$  ratio no greater than 0.32 by the year 2000.

## CONCLUSION

The results obtained in this study show that over the last 15 years  $^{239+240}\text{Pu}$ , and  $^{137}\text{Cs}$  concentrations in bivalves have decreased significantly. The new data, like the early results obtained in 1976-1978, show that  $^{241}\text{Am}$  activities are higher in samples from the West Coast than from the other locations. This has been related to the upwelling along the West coast of the intermediate Pacific waters associated with the California Current. In this study, it appears that a few spikes of activation products were detected in the vicinity of nuclear power plants.

## ACKNOWLEDGEMENTS

We are grateful to the Maryland Department of Natural Resources, Battelle, and Texas A & M University for their assistance in collecting and preparing the bivalves used in this study. Many thanks to Douglas Wolfe (NOAA) for providing useful information on the environmental behavior of radionuclides and for reviewing this manuscript and to Jeffrey Fritsen (Versar) for providing reference material. We also want to acknowledge Andrew Robertson and Thomas O'Connor (NOAA) for reviewing this manuscript.

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**TABLE CAPTIONS**

- TABLE 1** Location of the NS&T sites.
- TABLE 2** MDA and precision achieved for the radioisotopes measured in this project ( $\times 10^{-3}$  pCi).
- TABLE 3** Distribution of median, average and standard deviation of the radionuclides activities ( $\times 10^{-5}$  pCi/g, except for 40K pCi/g).
- TABLE 4** Comparison of the medians, averages and standard deviations calculated for NS&T and MWP70s data ( $\times 10^{-5}$  pCi/g).

## FIGURE CAPTIONS

**FIGURE 1:** Map showing the location of the NS&T sites where bivalves were collected for the Radionuclide Project.

**FIGURE 2:** Histograms showing the activity of radioisotopes in pCi/g displayed in a geographical order from the north of the East Coast through the Gulf of Mexico and ending in the Pacific Northwest.

- 2a Geographical distribution of  $^{241}\text{Am}$
- 2b Geographical distribution of  $^{137}\text{Cs}$
- 2c Geographical distribution of  $^{239+240}\text{Pu}$
- 2d Geographical distribution of  $^{238}\text{Pu}$

**FIGURE 3:** Histograms showing the activity of radioisotopes in pCi/g displayed in a geographical order from the north of the East Coast through the Gulf of Mexico and ending in the Pacific Northwest.

- 3a Geographical distribution of the isotope  $^{90}\text{Sr}$
- 3b Geographical distribution of the activation products
- 3c Geographical distribution of  $^{40}\text{K}$
- 3d Geographical distribution of  $^7\text{Be}$

**FIGURE 4** Histograms showing the geographical distribution of the activity pci/g of the radioisotopes for the MWP70s and for NS&T program

- 4a Comparison of NS&T and MWP70s data for  $^{241}\text{Am}$
- 4b Comparison of NS&T and MWP70s data for  $^{239+240}\text{Pu}$
- 4c Comparison of NS&T and MWP70s data for  $^{238}\text{Pu}$
- 4d Comparison of NS&T and MWP70s data for  $^{137}\text{Cs}$

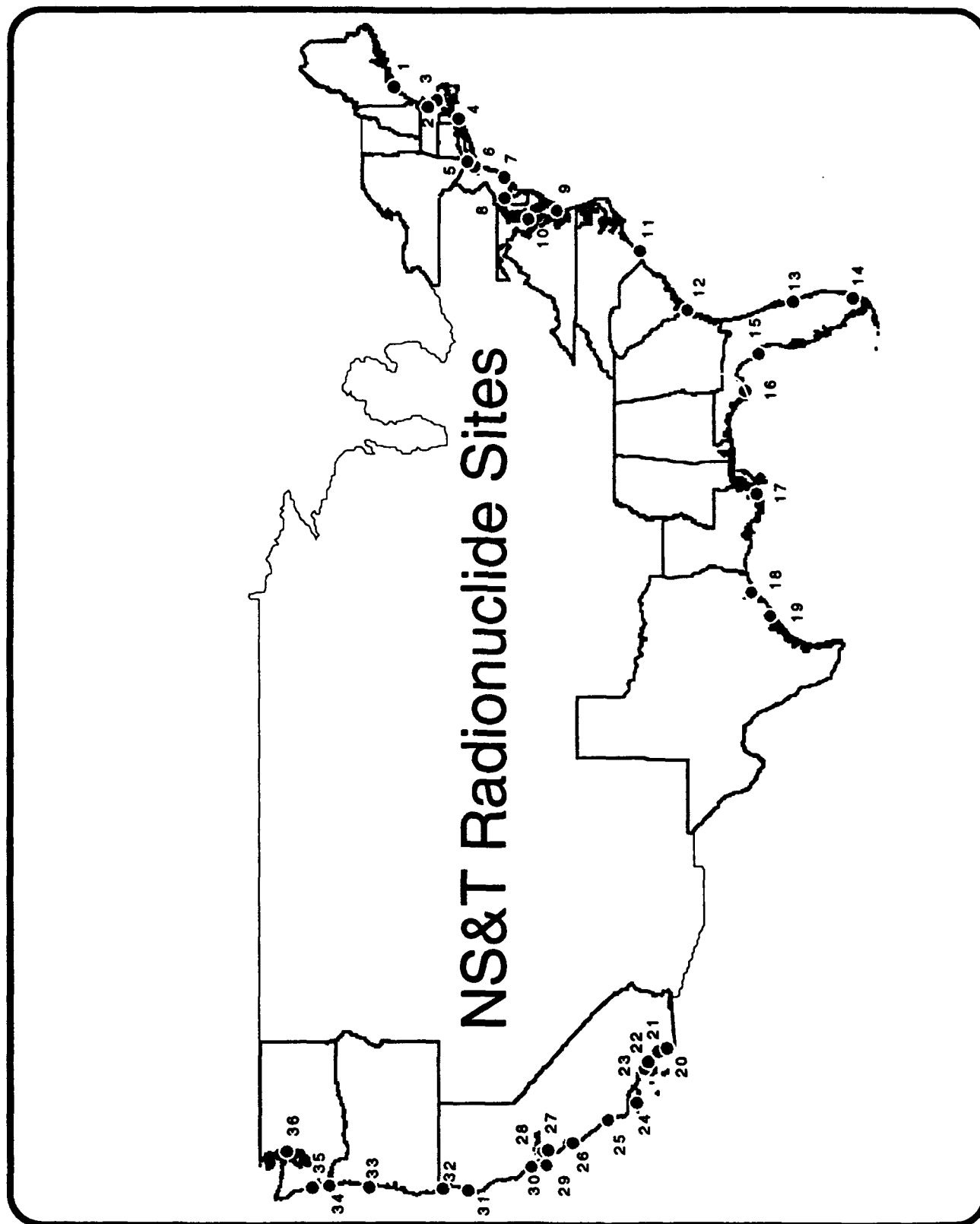
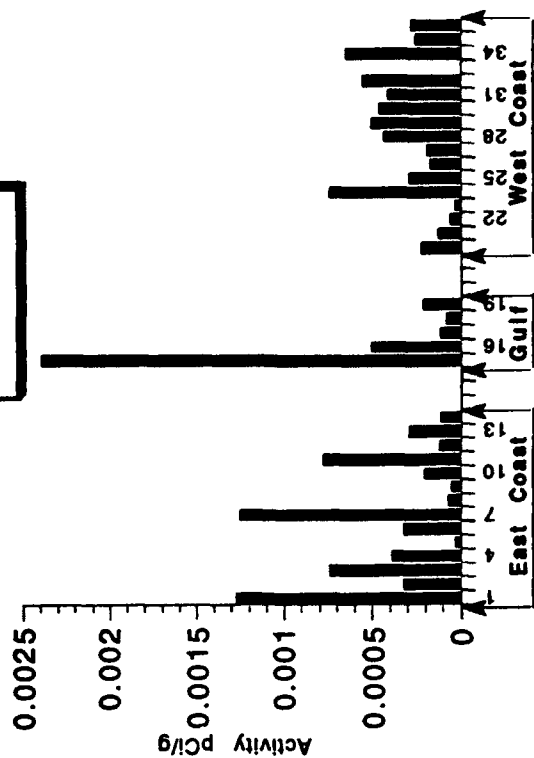


FIGURE 1

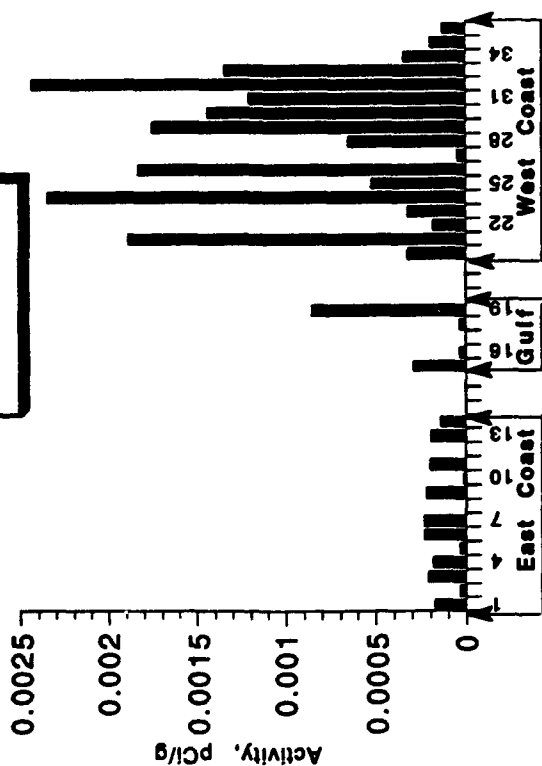


FIGURE 2

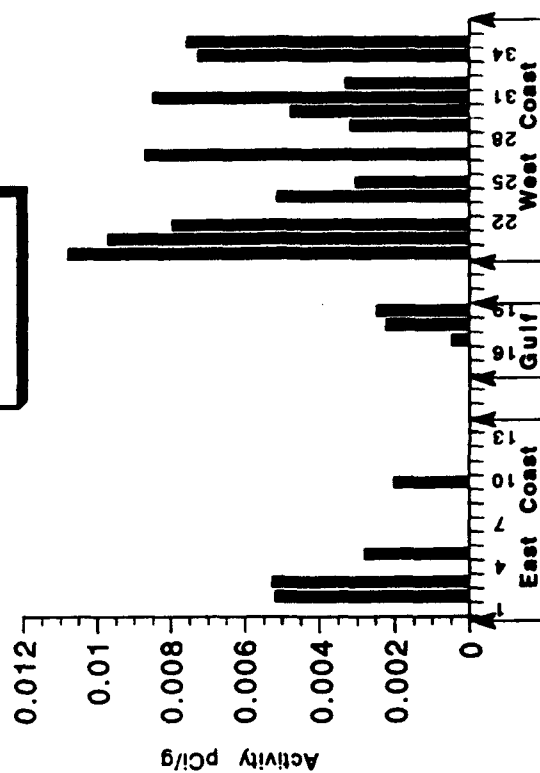
2 b

 $^{239+240}\text{Pu}$ 

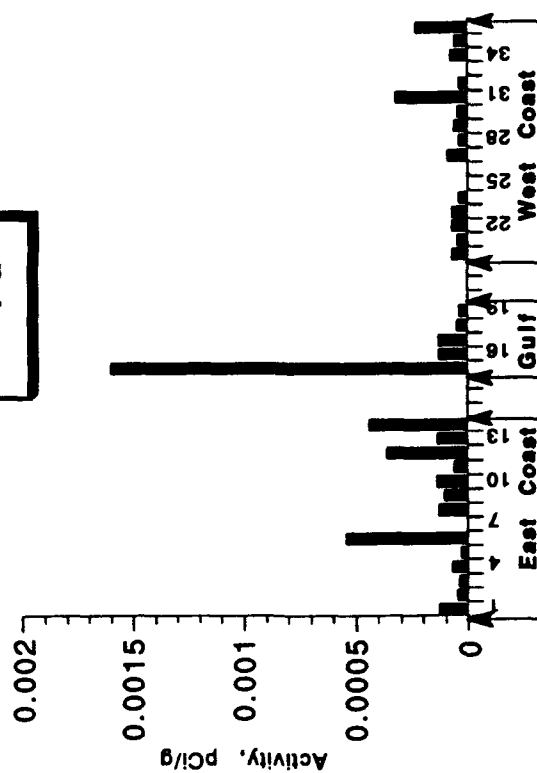
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 $^{241}\text{Am}$ 

2 d

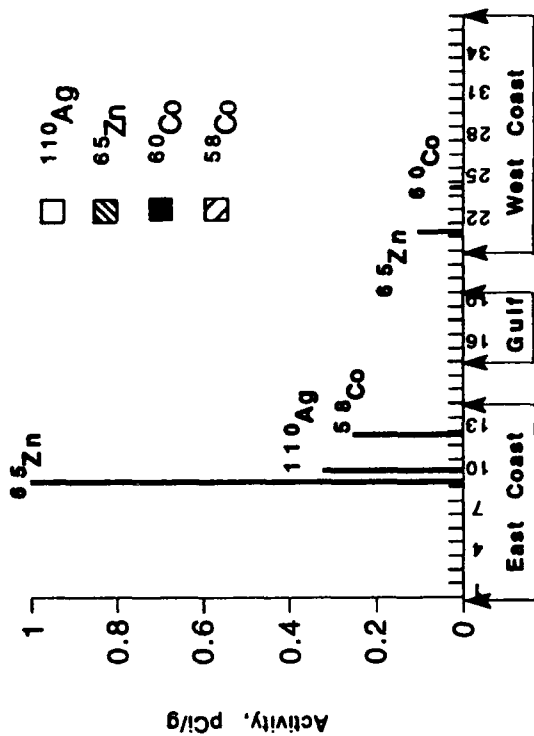
 $^{137}\text{Cs}$ 

2 c

 $^{238}\text{Pu}$ 

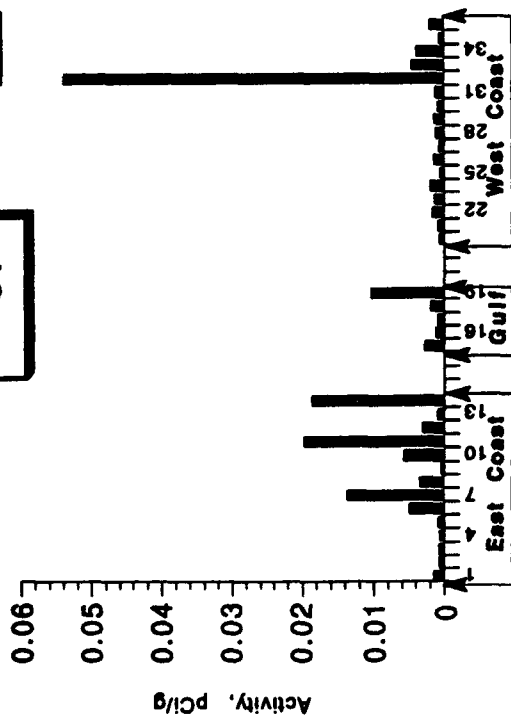
Activation Products

- ☐  $^{110}\text{Ag}$
- ☒  $^{65}\text{Zn}$
- ☒  $^{60}\text{Co}$
- ☒  $^{58}\text{Co}$



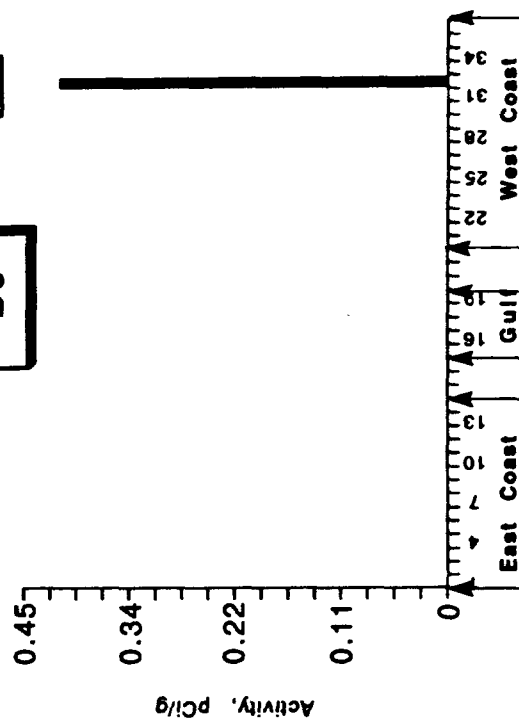
3a

$^{90}\text{Sr}$



3d

$^7\text{Be}$



3c

$^{40}\text{K}$

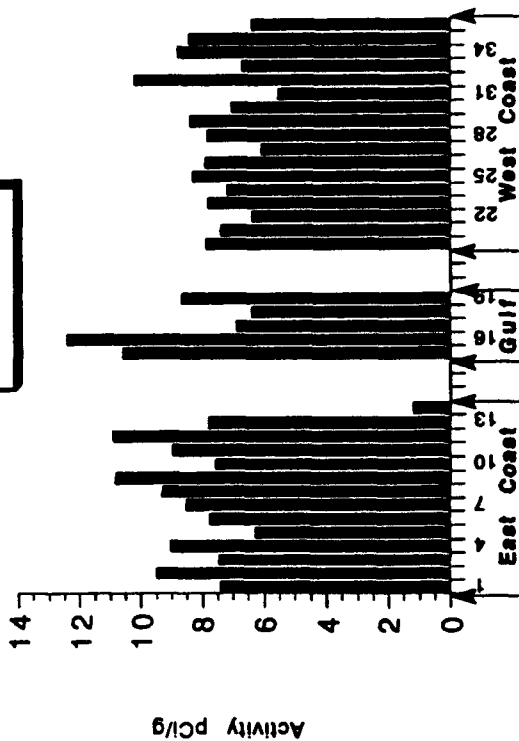
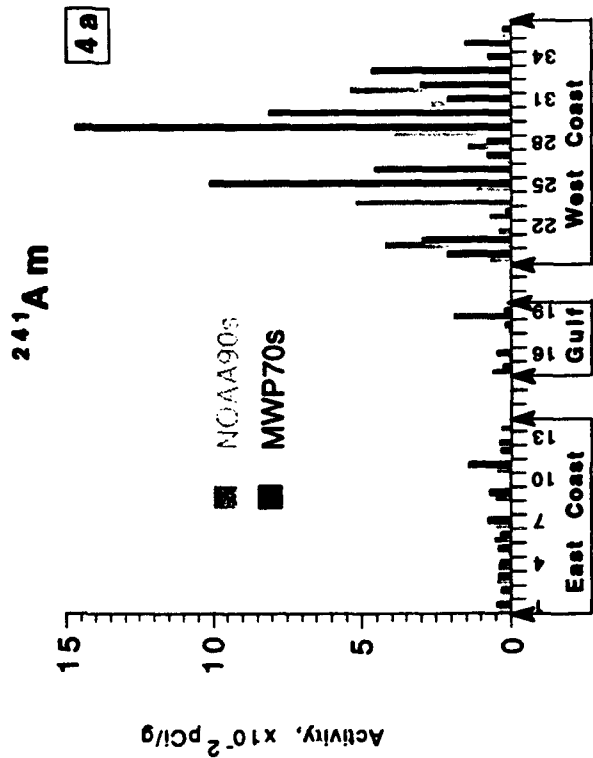
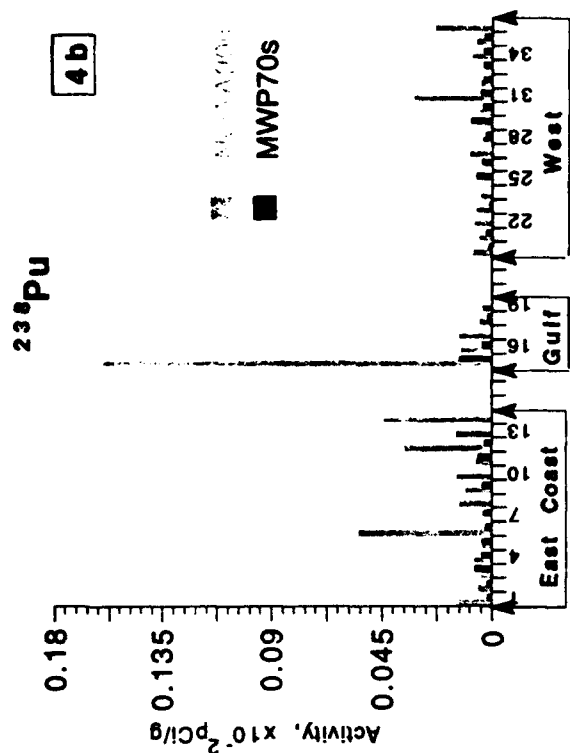


FIGURE 3

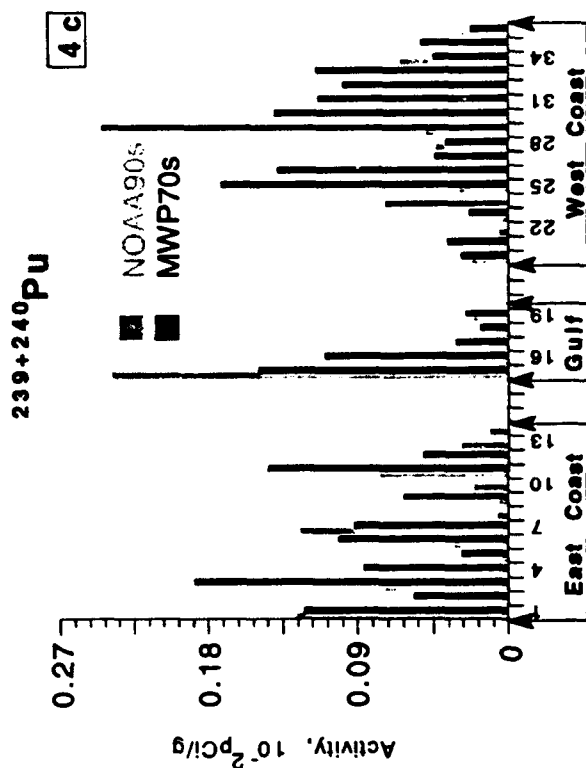
## Comparison NOAA/EPA Data



## Comparison NOAA/EPA Data



## Comparison NOAA/EPA Data



## Comparison NOAA/EPA Data

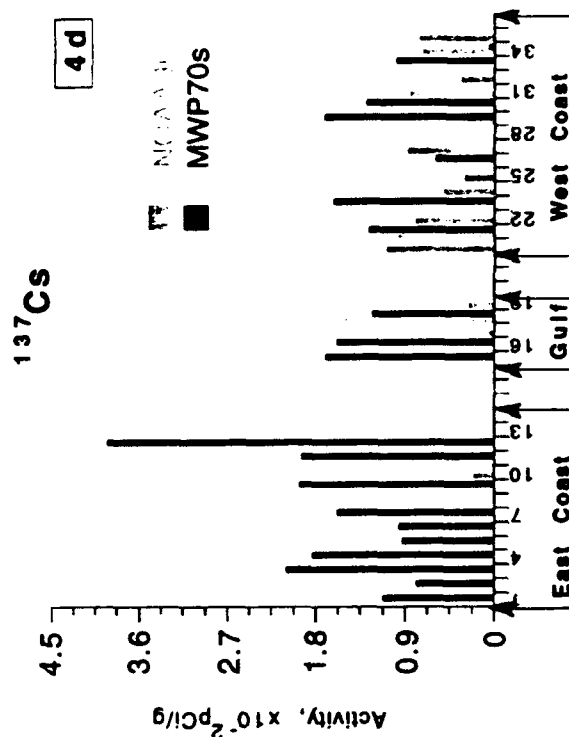


FIGURE 4



**Session 4: Legal, Economic and Policy  
Priorities**

Texts Accompanying Presentations

Prof. Yuri Barsegov  
Institute of World Economy and International Relations  
Russian Academy of Sciences, Moscow

CERTAIN LEGAL ISSUES CONCERNING PREVENTION  
OF RADIOACTIVE POLLUTION OF THE ARCTIC  
MARINE ENVIRONMENT FROM LAND-BASED SOURCES

Matters concerning the necessity of insuring international regulation of prevention of the radioactive contamination of the Arctic seas from land based sources are extremely acute. A great number of nuclear power plants which don't seem to be irreproachably safe and are notable for a high accident rate are concentrated on the Northern coasts and banks of rivers flowing into the Arctic waters. These are Sellafield, La Hague, Kyshtym and Tomsk, to name just a few. Such a reality became possible due to the fact that the international community failed to work out a universally fit and efficient legal mechanism for preventing pollution of such type.

Actually, there are no accepted rules, standards or procedures that could effectively ensure protection of the marine environment from harmful effects arising as a result of such activities. Even nowadays it is not the norms or principles of international conventions but rather ancient Roman maxims pertaining to private property (*sic utere tuo ut alienum non laedas*) that form the legal basis for imposing a ban on causing damage to the environment of other states or the environment of the open sea due to the activities under national jurisdiction.

By making references to certain judicial decisions scholars infer the forming of international legal norm stipulating that every state has an obligation not to allow its territory to be used

to the detriment of other states. Regretfully international norms are far less stringent when it comes to obligations of states to prevent or reduce environmental damage to areas outside their jurisdiction. The sole comprehensive legal instrument codifying existing norms pertaining marine environment protection is the UN Convention on the Law of the Sea. However, even it could not serve as an adequate legal basis for addressing issues of pollution from land-based sources. The Convention gives preference to solving such issues at national or regional levels.

It should be said in all fairness that the critics of the Convention are right when they say that it is very weak with respect to land-based pollution, that at the national, as well as international levels the Convention prescribes only vague, non-binding actions concerning pollution from land-based sources.

The Convention requires that the states in keeping with "the best practicable means at their disposal" and "in accordance with their capabilities" take all feasible measures on an individual basis as well as jointly with the aim of preventing, reducing and controlling pollution of the marine environment from any source. Among them are those designed to "minimize to the fullest possible extent" the release of toxic, harmful or noxious substances especially those that are persistent from land-based sources.

According to Article 207 of the Convention States have to adopt laws and regulations and take other measures as may be necessary, to prevent, reduce and control pollution of the marine environment from land-based sources "taking into account" of existing international rules, standards and recommended practices

and procedures. States are to "endeavor to harmonize" their relevant policies at appropriate regional levels. It is further stipulated that states acting through competent international organizations or diplomatic conference should "endeavor" to establish global and regional rules, standards and procedures taking into account characteristic regional features.

As regards Article 234 on "ice-covered areas" its provisions on the one hand contain acknowledgment of the fact that pollution of the marine environment in these areas "could cause major harm to or irreversible disturbance of the ecological balance" and on the other hand are confined to stating the rights of coastal states to adopt and enforce regulations for the prevention, reduction and control of marine pollution exclusively with respect to pollution from vessels.

Thus, although almost 80% of pollution is generated on land its control is left to the discretion of states.

The evident failure to adequately address the most pressing problems of marine pollution from land-based sources is due to a number of reasons.

Without doubt, global regulation of transboundary pollution of marine environment from land-based sources represents a substantially more complicated political and legal issue compared, for instance, with the international regulation of pollution from vessels or pollution caused by dumping.

It implies international environmental monitoring of economic activities within the territories of sovereign states. To a varying degree it presupposes an encroachment upon the activities, falling under the sovereign authority of each state within its territory.



This is something the states agree to with great reluctance and with numerous reservations especially when it comes to the operation of nuclear facilities to say nothing of secret military-industrial nuclear complexes.

That is why international regulation made certain headway only on regional levels. Regional regulation became the standing international practice that proceeds from the proximity of national interests of bordering states. Several regional conventions aimed at eliminating and controlling of land-based marine pollution have been adopted.

The first multilateral agreement to deal exclusively with land-based pollution was the 1974 Paris Convention for the Prevention of Marine pollution from land-based Sources in the North East Atlantic (entered into force 6.V. 1978).

Prior to the adoption of the 1982 Convention on the Law of the Sea relevant instrument had also been adopted for the Baltic Sea (the 1974 Helsinki Convention) and the Mediterranean (the 1980 Athens Protocol for the Protection of the Mediterranean Sea Area against Pollution from the land-based Sources). Similar agreements have since been adopted for the South-East Pacific (The 1983 Quito Protocol for the Protection of the South-East Pacific Against Pollution from Land-based Sources), the Persian Gulf region (the 1990 Kuwait Protocol on the Protection of the Marine Environment against Pollution from Land-based Sources), the Black Sea (the 1992 Convention on Protection of the Black Sea against Pollution, its Protocols and Resolutions).

Work is currently underway on similar instruments for the

Caribbean region, as well as Western and Central Africa.

States-parties to the aforementioned agreements assumed obligations with respect to eliminating and monitoring land-based pollution of respective marine areas. In particular, states-parties to the 1974 Paris Convention undertook an obligation to prevent and within the limits of the possible eliminate radioactive pollution of the Arctic region. The scope of these agreements is limited and doesn't cover wide areas of the Arctic Region. Only a small part of it - areas of the Atlantic and the Arctic Oceans located 36 Parallel between longitude 42° West and 51° East are covered by 1974 Paris Convention. Besides, such an Arctic state as the USSR wasn't a party to it.

Issues concerning the greater part of the Arctic region are still not addressed to. Moreover the provisions of the Paris Convention as well as of other regional instruments were not patterned on the adoption of drastic measures. They were essentially aimed at banning discharges of high and medium-level liquid radioactive waste products rather than preventing the most widespread discharge of low-level radioactive cooling water and other effluents from nuclear-power plants and industrial works into inland hydrosystems communicating with the seas or directly into sea.

The fact that the Paris Convention as well as other regional agreements accepted the principle of the most advanced technologies for determining acceptable levels of discharges as including those which are insignificant and close to natural levels practically legalized direct discharges of low level radioactive effluents into the environment in gaseous and liquid form from nuclear

power-plants and works operating in the regions covered by the aforementioned agreements.

Practically speaking all legal means of preventing radioactive pollution of the marine environment from land-based sources are focused on national legislation. Although municipal legislation is aimed not so much at carrying out relevant treaty provisions as at national environment-conservation regulation it nevertheless opens certain possibilities for giving effect to existing norms, standards and recommended practices by way of their incorporation into relevant municipal legal acts. Such incorporation of international norms into national laws is left to the discretion of states and hence is subject to restrictive interpretations. National legal acts very frequently weaken international norms. This is so because states in fact agree only with those basic requirements that if not observed would in fact make activities of states inconsistent with the aims and norms of the treaties.

As to the Arctic region even that kind of mechanism is lacking. Actually there are no legally binding standards concerning prevention of the pollution from land-based sources whatsoever and the state of affairs in the area is determined exclusively by economic and technological development of the Arctic basin countries which have capabilities in the field of nuclear energy, by their national priorities as well as the level of ecological awareness.

This can be clearly seen if we compare levels of radioactive pollution of marine areas adjacent to nuclear fuel processing plants of Britain (Sellafield) and France (La Hague). These two

countries also stand relatively close with respect to scientific and technological development. The level of beta-radiation and alpha-radiation pollution produced from the Sellafield plant in 1975-79 was respectively 3,5 and 75 times higher than the rate of pollution at the La Hague nuclear-plant. Although today the gap has been narrowed nevertheless the British plant still has a higher pollution rate.

One can conclude that the current practice of international legal regulation of measures pertaining to prevention of radioactive pollution of the environment from the land-based sources doesn't correspond to the present day requirements. This is even more so in respect to the Arctic - the region so vital and yet so vulnerable.

Probably it is premature so far to speak either of any unification of respective legal norms or search for comprehensive global solutions. Evidently, under existing conditions, it is extremely difficult to take drastic measures on preventing radioactive pollution of the Arctic basin by working out rigid nuclear safety regulatory regime.

Probably the "Rovaniemi Process" has become the most effective and representative institutional mechanism of intergovernmental cooperation in the Arctic region. A well-known Declaration on the Arctic Environment protection and the Strategy of Arctic Environment were adopted within its framework. These two documents singled out issues of preventing radioactive pollution of Arctic as a special objective. Nevertheless this mechanism was set up not for solving but rather for discussing and analyzing the problem. The chances of an early adoption on its basis of coordinated decisions

of legal character specifically on the prevention of radioactive pollution of the Arctic from land-based sources are slight.

The most realistic thing to do would be to broaden and expand existing and work out new programs of monitoring and to give incentive to international cooperation in the conduct of joint radiological research on the regional as well as global level.

There is also an issue of adopting a "precautionary approach" regarding the protection of the Arctic marine environment from land-based radioactive pollution. The precautionary principle is being widely accepted in environmental practice.

As a general principle to be followed by states in preserving the environment it was incorporated into the 1992 Maastricht Treaty of the European Union.

The 1992 Convention on the protection of the marine environment of the Baltic Sea area clearly defines the precautionary principle in respect of hazardous effects of persistent toxic and bioaccumulative substances. The 1992 Convention on the Protection of the Marine Environment of the North-Eastern Atlantic and on the Protection of the Marine Environment of the Black Sea also contain this principle.

Precautionary approach appears also in the Final Declaration (Plan of Actions - "Agenda XXI") adopted by the UN Conference on Environment and Development.

Some specialists in the field of environmental law consider the precautionary principle as being a higher, and therefore, guiding or fundamental legal principle.

However, despite the fact that the term has appeared with

increasing frequency in several international instruments (mostly regional agreements), the "precautionary principle" can hardly be regarded as a general principle of international law, legally binding all states.

However, adoption of this emerging principle by all states of the Arctic basin with regard to radioactive pollution of the Arctic seas could serve as a considerable contribution to the progressive development of the international environmental law.

For the time being, due to the complexity of elaborating concerted actions on the prevention of radioactive pollution of the Arctic from the land-based sources, the prospects of introducing this principle depend not so much on the signing of an international multilateral convention as on its incorporation in domestic laws of the Arctic states.

The recent environmental legal act adopted in Germany can serve as an example of application of the precautionary principle in municipal law.

The adoption of the principle internationally or by national legislation of the Arctic states would mean their willingness, before undertaking any activity that may adversely affect the quality of the environment, to assess the impact of such activity on the environment. In its broadest sense the principle has been described as an international obligation to prevent pollution. In its maximized interpretation the precautionary principle urges that substances or activities that may be harmful to the environment should be regulated even if conclusive scientific evidence of their harmfulness is not available.

It's not all that important whether such a consent is expressed in the form of a multilateral convention or national legislation. It's far more essential that all states on which the prevention of land-based radioactive pollution of the Arctic depends, agree to the principle.

The incorporation of the principle into national environmental law of the Arctic states doesn't provide for any interference into the jurisdiction and sovereign authority of states. Such encroachment upon the prerogatives of states would be almost inevitable when the principle falls under international legal order. National regulation makes the solution easier and more realistic. Reflection of this principle in relevant national laws of the Arctic basin states could be done in various ways. The availability of such options opens the possibility to press individually for setting internationally accepted requirements and standards for the enterprises which are related to nuclear power and fission products.

In the long run application of this principle would induce states to perfect existing nuclear technologies and introduce advanced, practically close-circled nuclear installations recommended by the UN Committee on the Effects of Nuclear Radiation as a model with respect to ensuring environmental security.

All this would pave the way for appropriate international legal steps on regional and, perhaps, global scale.

In the broad context of establishing legal basis for preventing radioactive pollution of the Arctic waters one cannot but touch upon a subject of liability and responsibility of states

for such kind of pollution.

The aforementioned Treaty of the European Union considers liability as being part of a general principle of preventing pollution of the environment. Article 130 of the Treaty explicitly states that the European policy of environmental protection is based on assumption that environmental damage should as a priority be rectified at source and that polluter should pay.

It is evident that the adequate settlement of issues of liability for harm caused to the environment (including the marine environment) would be the most efficient way of protecting it.

This issue has been on the agenda of the UN International Law Commission for many years. Regretfully the work is still very far from completion. This is so due to the great difficulties both practical as well as theoretical (conceptual).

In order to get a precise notion of the legal nature of liability one had to separate the liability, i.e. no-fault responsibility, from responsibility for wrongful acts, for breaches of international obligations.

International law should provide for the responsibility of states for intentionally inflicted damage to the environment of another state within its territory, zones of national jurisdiction or areas beyond their limits. Personal responsibility of natural persons, government agents and even members of the government should also be envisaged with respect to environmental crimes - acts committed intentionally and on a large scale. Such crimes should be qualified as crimes against humanity, since damage inflicted on the environment of the state and even more so of the



areas open to all states, affects the whole of the mankind.

Due to specific character of radioactive contamination pollution by one state of the territory of another state or of the high seas from land-based sources is hardly possible: its destructive effect would inflict damage first of all on its own territory and its own marine areas (territorial sea, exclusive economic zone, continental shelf). Nevertheless, pollution of that kind is quite possible as a result of culpable negligence. In this case responsibility would be enforced by national institutions of justice.

Responsibility resulting from wrongful acts doesn't exclude the possibility and necessity of compensating the harm or damage. But the legal nature of this obligation is fundamentally different from that arising from a strict of objective liability.

The matter of a strict or objective liability for transboundary damage caused by activities conducted in conformity with the international law appears to be more complicated.

The establishment of international rules governing strict or objective liability was complicated since the existing treaty regulation was somewhat inadequate and since there was not enough evidence of a general practice accepted as customary rules and principles which could serve as a basis for general and universal concept of liability.

Following years of efforts full of attempts, errors and improvisations the International Law Commission seems to succeed in sketching out a concept of liability to be incorporated into public international law.

FRAMEWORK FOR A REGIONAL ENVIRONMENTAL REGIME FOR THE ARCTIC

by

Aldo Chircop

(Presented at the First International Conference on Radioactivity and Environmental Security in the Oceans: New Research and Policy Priorities in the Arctic and North Atlantic, Woods Hole Oceanographic Institution, 7-9 June 1993)

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### Introduction

There has been reference in this conference to the Arctic as a type of Mediterranean, a middle sea or a semi-enclosed sea. The United Nations Convention on the Law of the Sea 1982 (LOS Convention) proposes a cooperative ethic for states bordering an enclosed or semi-enclosed sea, which is defined as "a gulf or basin surrounded by two or more states and connected to another sea or the ocean by a narrow outlet or consisting entirely or primarily of the territorial seas or EEZs of two or more coastal states."<sup>1</sup>

In particular, the ethic translates into cooperation in the exercise of the rights and performance of duties under the LOS Convention. These rights and duties refer as much to marine resource development and general ocean use as to the protection and preservation of the marine environment, marine scientific research and transfer of technology.<sup>2</sup> These issues are all relevant to environmental regime-building in the Arctic.

There are now a number of models for regional environmental regimes, such as those in the eleven regions of the Regional Seas Programme of the United Nations Environment Program (UNEP), the North Sea, the Baltic, the Black Sea, the Gulf of Maine and the Antarctic. They all involve a political commitment by the parties adhering to them.

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<sup>1</sup>. Article 122, United Nations Convention on the Law of the Sea, A/Conf.62/122, 7 October 1982.

<sup>2</sup> Article 123, ibid.

The UNEP Regional Seas Program has by far been the most influential and has in fact influenced the development of the other regimes. If the "Mediterranean model" is used, then what is contemplated is an action plan with an overall integrated management approach, having four main components addressing integrated planning and management, scientific research and monitoring, a legal framework, and an institutional and financial basis.

#### Integrated management component

Integrated management would serve as an overall policy. This approach involves looking at the Arctic as a whole, indeed as a system within which the actions of some in one area are likely to affect the well-being and value pursuits of others in other areas. There are also socio-economic and cultural considerations concerning the peoples of the Arctic and management of the coastal zone. Since the Arctic would be considered as a system, then both areas within national jurisdictions and the high seas in between should be included in the regime's geographical coverage.

#### Scientific component

This conference is a good example as to how this component could be developed. Coordinated marine scientific research, assessment and monitoring will provide the essential knowledge base of the regime. This is a necessary intelligence function to facilitate informed, efficient and relevant decision-making.

Legal component

This component is an expression of a basic duty at international law to protect and preserve the marine environment. The legal component provides a normative and regulatory framework for activities that may be deemed to be harmful or carry risk for the Arctic environment. An important provision in the LOS Convention already provides a basis for coastal states to opt for stricter regulation of activities in ice-covered areas within national jurisdiction.<sup>3</sup>

Although there are already a number of global environmental law conventions, the unique environment of this region suggests a problem-oriented and contextual approach. More concretely, a look at other regional environmental regimes suggests the development and adoption of a framework "Convention for the Protection of the Arctic Environment," and at least two protocols dealing with land-based sources of pollution and dumping. These instruments should include all waters and the seabed, including internal waters enclosed by the straight baseline systems in the region. These instruments should be guided by the precautionary approach, the duties to inform and assist, the duty to cooperate, and efforts should be made to develop a suitable responsibility and liability regime. Specifically in relation to environmental responsibility, four matters would need to be addressed: attribution of

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<sup>3</sup> Article 234, ibid.

responsibility; nature of liability; assessment of damage/harm; and remedies.

#### Institutional component

The degree of formalisation of this regime and the type of structure to service it are political decisions. The UNEP Regional Seas Programme has treaties and elaborate institutional structures in the various regions,<sup>4</sup> whereas the Gulf of Maine Action Plan is an agreement between three US states and two Canadian provinces working with a simple informal structure.<sup>5</sup> In the Arctic it would be desirable to have some type of structure that would ensure regular official meetings of regional states, other interested states and stakeholder groups.

As in the case of other regional environmental regimes financial arrangements would be needed, since costs will be incurred and should be shared equitably or in a politically acceptable manner.

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<sup>4</sup> The most complex is the institutional framework of the Mediterranean Action Plan. The principal organ is the biennial Meeting of the Contracting Parties. A Bureau works closely with the Coordinating Unit, whose responsibilities include the coordination of a number of regional activity centres. Each centre has a network of national focal points. For a detailed study of this institutional arrangement, see Aldo Chircop, Cooperative Regimes in Ocean Management: A Study in Mediterranean Regionalism, JSD dissertation, Dalhousie University, Halifax N.S., 255-372.

<sup>5</sup> The main organ is the Gulf of Maine Council which is composed of the state/provincial environment and environment-related ministries. The Working Group is the principal body through which the action plan is implemented, and consists of senior state/provincial government members, federal government representatives, and more recently, private sector representatives.

Finally, environmental regimes need the support of public and stakeholder constituencies to ensure workability in the long-term. This is an element that should be developed at an early stage of regime-building, as non-involvement or non-meaningful involvement could easily undermine the acceptability of a regime by stakeholders that are supposed to benefit from it.

Time prevents me from elaborating on these issues and expected advantages and disadvantages of the various approaches in an Arctic context. Suffice to note that regime-building is as complex as the environment addressed and the political demands for cooperation expressed.

## **The London Convention and Radioactive Waste Dumping at Sea: A Global Treaty Regime in Transition**

Prepared by Clifton Curtis \* for the  
Conference on "Radioactivity and Environmental Security in the Oceans:  
New Research and Policy Priorities in the Arctic and North Atlantic"  
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### **I. Introduction**

At their 15th Consultative Meeting, in November 1992, the Parties to the Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter (1972), commonly known until then as the London Dumping Convention (LDC), decided to change their informal name to the London Convention (LC). In doing so, they sent a message to the international community, following the Earth Summit, that the time had come to move beyond a regime governing sea dumping, toward a precautionary approach focused on protecting the marine environment in an ecologically sound manner.

As of June, 1993, however, dumping at sea of intermediate- and low-level radioactive wastes still is not legally banned. The dumping of such wastes has been at the center of a lengthy international political and legal environmental controversy dating back to the late 1970s. This controversy is expected to culminate, however, with changes to the LC at a special "Amendment Conference" scheduled for November 1994. In this regard, most States believe that radioactive waste dumping at sea is out-of-step with current trends and state-of-the-art contemporary environmental policy, and they will sponsor or support a formal ban next year, yet a few powerful countries remain reluctant about the prospect of a such a global, legally binding prohibition.

### **II. Historical Context**

The deliberate dumping of wastes at sea, from ships, aircraft, platforms and other man-made structures is regulated worldwide by the LC, which entered into force in 1975. The LC was adopted in late 1972, shortly after the UN Stockholm Conference on the Human Environment. With 70 Contracting Parties to date [1], the LC is one of the main global treaties for the prevention of marine pollution. Its basic purpose is to encourage nations of the world to work together to ensure that the marine environment is protected from the hazards of dumping, with "protection of the marine environment", broadly defined, as its foundation. In this regard, Article I provides that:

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"Contracting Parties shall individually and collectively promote the effective control of all sources of pollution of the marine environment."

The LC contains three annexes: Annex I, the so-called "black list", includes the substances for which dumping at sea is strictly prohibited; Annex II lists substances for which dumping requires the issuance of special permits; and Annex III establishes criteria governing issuance of dumping permits. The LC and its annexes can be amended by two-thirds majority vote of those present at meetings. Annual consultative meetings are held in London at the International Maritime Organization (IMO), the specialized UN agency responsible for LC secretariat duties. Originally, LC meetings were designed to do little more than collect (incomplete) data on and "regulate" dumping activities. But since the early 1980s, those meetings have had increasing political importance, in line with political and social demands for clean seas and a cleaner environment. This growing political profile became especially evident as the global debate surrounding ocean dumping of radioactive wastes took place under the LC's auspices.

### III. The Radioactive Waste Dumping Controversy

When the LC was adopted in 1972, high-level radioactive wastes (HLW) were placed on the Convention's "black list" (Annex I), but intermediate- and low-level radioactive wastes (ILW and LLW) were placed on Annex II in order to allow the regulated disposal at sea of these inconvenient wastes. However, by 1983 the dumping of radioactive wastes at sea was no longer considered to be an acceptable option for the vast majority of public opinion worldwide, and -- as a result -- for the majority of LC Parties. Given concerns over possible environmental consequences of dumping, including impacts on other legitimate uses of the sea such as fisheries and tourism, three separate, but related proposals were presented at the LC's 7th (1983) Consultative Meeting.

First, two small island nations from the South Pacific, the Republics of Nauru and Kiribati, concerned by Japanese and US plans to resume sea dumping on a vast scale in the Pacific ocean [2], put forward a proposal to amend the Convention so that all radioactive wastes (ILW and LLW in addition to HLW) would be placed on Annex I's "black list". Second, the five Nordic countries (Denmark, Finland, Iceland, Norway and Sweden) presented a proposal by which radioactive waste dumping at sea would be "phased out" by 1990. This proposal was prompted by concern about the rapid increase in the quantities of radioactive wastes which had been dumped at sea in the early 1980s.[3] Third, the government of Spain - severely affected by the proximity of several dumpsites (in particular the NE Atlantic dumpsite used by eight member States of the Nuclear Energy Agency (NEA) of the OECD between 1967 and 1982) [4] -- called for an immediate moratorium on all radioactive waste dumping operations. Summarily stated, the objections to sea dumping underlying those three initiatives were that:

- 1) the oceans are a living, interconnected environment that can return radioactive wastes to humans via ocean food chains. The objective of radioactive waste management

should be to contain those wastes so that they are isolated from the biosphere instead of diluting and dispersing them in the environment, which is the case with ocean dumping;

2) the ocean is a formidable environment. Pressures and temperatures reach there planetary extremes and the corrosive powers of ocean waters are legendary. The ocean can be a very destructive environment for placement of radioactive waste containers;

3) the oceans are still relatively unknown. New discoveries surface regularly. Scientific opinion on the capacity of the seas to absorb wastes is rapidly evolving;

4) the oceans represent a "global commons" which should be preserved for the benefit of all people and future generations. It is fundamentally unfair for a minority of the planet's population to disproportionately damage shared marine resources and thus deprive the vast majority of their own rights; and

5) those States which were dumping radioactive wastes at sea were not willing to accept the burden of proving the safety of such a practice -- instead preferring the much easier, more traditional, approach which requires that such a burden rests with those who need to prove that such a practice is harmful.

The "South Pacific" and "Nordic" proposals were strategic in nature: they were both tabled in order to allow the passage -- after a long, acrimonious debate -- of the Spanish proposal, establishing a moratorium pending the outcome of a meeting of experts on radioactive wastes.[5] Resolution LDC 14(7), by which the moratorium was established, was adopted by a vote of 19 in favor, 6 against and 5 abstentions. (See attached Table for breakdown of voting.)

Resolution LDC 14(7) was legally non-binding, and the UK and certain other OECD countries indicated plans to ignore it, and proceed with dumping at sea in mid-1983. But the dump ships were forced to stay in port, largely due to the strength of political pressure created by public opinion and trade unions from all over Western Europe. That campaign culminated in the summer of 1983 when the National Union of Seamen (NUS) of the UK and the International Transport Federation (ITF) -- with affiliates worldwide - - called upon its workers not to handle radioactive wastes destined for sea dumping. Their call was heard: the workers did not cooperate, and planned dumping cruises were canceled. Instead, the radioactive wastes were stored on land.

In 1985, a report by a panel of "experts" appointed by the IAEA and ICSU [6] was produced. In response, the 9th (1985) Consultative Meeting had to decide whether or not ocean dumping could be resumed. However, numerous delegations felt that the IAEA/ICSU participants in the panel of experts tended, with some exceptions, to be proponents of an obsolescent, but nonetheless influential scientific perspective that the oceans' "assimilative capacity" to receive humankind's wastes and other hazardous substances remained limitless. Unsatisfied by the panel's response, the Spanish

government proposed another resolution suspending "all dumping at sea of radioactive wastes and other radioactive matter" in order to "permit time for... a broader basis for an informed judgement on proposals" for amending the LC, and requesting that "additional studies and assessments of the wider political, legal, economic and social aspects of radioactive waste dumping at sea be undertaken ...."[7] That Resolution LDC.21(9), establishing a moratorium for an indefinite period, which is still in force, was adopted by an overwhelming majority of 26 in favor, 5 against and 7 abstentions. (See Table)

After the 1985 moratorium, the LC Parties established an Intergovernmental Panel of Experts on Radioactive Wastes (IGPRAD) to address some of the questions raised by the resolution. [8] The IAEA was also asked to come up with a series of reports on other issues, including a complete inventory of all radioactive wastes entering the marine environment, and a comparative study of sea dumping with land-based options. IGPRAD has held five sessions since 1988, and will hold a final one from 12-16 July, 1993, to finish its report for presentation to the 16th (November, 1993) Consultative Meeting.

#### IV. Future Options

In light of the substantive differences of opinion between the very small minority of States who want to keep open the option of dumping radioactive wastes at sea (principally Japan, the UK, France, and the US), and the majority anti-dumping States, IGPRAD has considered a list of options that will be included in its final report. It is clear that IGPRAD has "essentially accepted that this range of options should be presented in the final report, with some discussion of their merits and demerits but without a recommendation to the Consultative Meeting necessarily being made".[9]

The range of options which are being considered for inclusion in the final report of IGPRAD have been summarized as follows [10]: Option 1 -- lift the moratorium, allowing sea disposal of LLW in accordance with existing LC and IAEA rules; Option 2 -- lift the moratorium, allowing sea disposal in accordance with strengthened international rules. This could include revised LC and IAEA rules and any additional guidance agreed to by contracting parties; Option 3 -- link action on radioactive waste to the resolution on a phase-out of industrial waste dumping adopted at the 13th Consultative Meeting, by which the dumping of such waste is to be phased out by the end of 1995; Option 4 -- continue a time-specific or indefinite moratorium; Option 5 -- develop new special consultative procedure governing the sea disposal of radioactive waste; Option 6 -- prohibit disposal of radioactive waste by amending the LC; and Option 7 -- prohibit disposal of radioactive waste by amending the LC, with an "opting out" possibility after a certain agreed upon time, under certain agreed upon conditions, for certain countries who might not be in a position to readily accept an outright immediate ban.

It appears that a resumption of ocean dumping as described in options 1 and 2 would be out of step with the current trend of international environmental policy, in particular with the precautionary approach which has become the guiding principle in many

international fora, including the LC, and which was endorsed in Principle 15 of the Rio Declaration on Environment and Development. In addition, it also appears that those options would conflict with the Earth Summit's Agenda 21.[11] Indeed, Agenda 21's Chapter 22, entitled "Promoting the Safe and Environmentally Sound Management of Radioactive Wastes", includes language directly relevant to the issue of radioactive waste dumping under the LC, calling upon States to (para. 22.5(b)):

Encourage the London Dumping Convention to expedite work to complete studies on replacing the current voluntary moratorium on disposal of low-level radioactive wastes at sea by a ban, taking into account the precautionary approach, with a view to taking a well informed and timely decision on the issue.

While this language does not specifically call for an immediate ban on radioactive waste dumping, it is clearly indicative of the global trend against the dumping of radioactive wastes at sea. In particular, it is interesting to note that the purpose acknowledged for the IGPRAD studies is "to replace the current voluntary moratorium...by a ban". Agenda 21 nowhere mentions the possibility of a resumption of dumping, and by not including such an option, the Heads of States at the Earth Summit sent a clear message to the LC Parties. Paragraph 22.5(c) of Agenda 21 also is directly relevant, admonishing States to:

[n]ot promote or allow the storage or disposal of high-level, intermediate-level and low-level radioactive wastes near the marine environment unless they determine that the scientific evidence, consistent with the applicable internationally agreed principles and guidelines, shows that such storage or disposal poses no unacceptable risk to people and the marine environment or does not interfere with other legitimate uses of the sea, making, in the process of consideration, appropriate use of the concept of the precautionary principle.

It should be clear that if States are not to promote or allow disposal near the marine environment, absent compliance with the stringent criteria contained in paragraph 22.5(c), then they certainly should not do so for disposal in the marine environment. Any other interpretation, especially when read in combination with paragraph 22.5(b) would effectively constitute a perversion of the spirit of Rio.

Option 4 of the IGPRAD proposals, continuation of a moratorium (time-specific or indefinite) would merely amount to a continuance of the status quo. Ten years after the first resolution establishing the moratorium was adopted, and after a decade-long thorough review showing that the nuclear industry's case in favor of sea dumping could not technically be justified, the reestablishment of a moratorium would be perceived as a very limited advancement. Technically, although no one (besides the former USSR) has officially violated the moratorium, it is still non-binding, and the threat of a resumption

of this practice would remain. This concern does not preclude the possibility that Resolution LDC.21(9) now reflects a rule of customary international law.

Option 3 would link action on radioactive waste to Resolution LDC.43(13) adopted at the 13th (1990) Consultative Meeting, and by which the Parties agreed "that the dumping of industrial wastes shall cease by 31 December 1995 at the latest". For the purpose of that resolution, "industrial wastes" was defined as "waste materials generated by manufacturing or processing operations", and it was agreed that "the inclusion of radioactive matter in this definition would be considered when the current London Dumping Convention review of issues relating to radioactive waste dumping has been completed".[12] In light of the inconclusive IGPRAD report, a permanent ban on radioactive waste dumping at sea would be consistent with Resolution LDC.43(13), as long as that resolution is effectively and properly complied with and enforced.

Under Option 5, "Contracting Parties would not allow sea disposal, except as carried out under a new special procedure agreed by them. The new procedure could be elaborated in an addendum to Annex I, a new annex or protocol, or a new section in Annex II." [13] A nuclear accident causing contamination of considerable areas of land is a case that was mentioned, informally, by some IGPRAD participants, where sea dumping might be an "option of least detriment". Although such an approach may have some merits, it is questionable whether it could realistically be implemented. For example, it is doubtful whether "pro-dumping" governments would agree to a decision making procedure that would provide that dumping operations could only be allowed on a case-by-case basis with the consensus of all LC Parties. There also are fears that there might be different interpretations of what constitutes a special or an extraordinary circumstance.[14]

In light of these considerations, among others, it appears that the most reasonable and broadly acceptable course of action is that presented by Option 6 -- to prohibit disposal of radioactive waste by amending the LC. Indeed, ocean dumping of radioactive wastes has been the subject of decade-long critical review, yet the nuclear industry has failed to convince the vast majority of Parties to the LC that there are valid reasons to allow the resumption of this method of disposal. It has even been said that, had the ocean dumping controversy started today, rather than the 1980s, the adoption of a permanent ban would have been much faster, and the terms of reference for and composition of IGPRAD would have been very different.

#### **Special Considerations Concerning De-Minimis and Entry into Force**

Beside substantive objections, some -- in particular the US delegation -- have raised two concerns with regard Option 6: the need to establish de-minimis levels of radioactivity below which dumping could take place; and the fact that even if the Parties decide this year to amend the LC, these amendments are unlikely to enter into force for several years, in light of the inherently slow process of ratification. Neither concern, however, provides a reasonable basis for postponing action on amendments.

Concern over de-minimis levels is based on the fact that -- strictly speaking -- "everything" is radioactive, and that, as a result, a level of radioactivity "below regulatory concern" should be established before the dumping at sea of "radioactive waste and other matter" is banned. Paradoxically, advocates of a de-minimis concept who base their reasoning on the need for clarity may well be complicating a simple issue: it is well known what is and is not commonly known as a "radioactive waste". In fact, the LC moratorium has worked well without requiring a de-minimis definition, and -- likewise -- several regional legal instruments have banned radioactive waste dumping at sea without such definitions. [15] Certainly a de-minimis definition would be desirable in relation to a ban on radioactive waste dumping, since it would prevent Parties from exploiting loopholes via lax domestic definitions. Efforts to agree on such a definition could be spelled out in the future, though, and need not delay amendment action.

Regarding entry into force, it is true that LC amendments, adopted by a two-thirds majority of those present, are not binding on Parties which have not accepted them, and that others who did not participate in the decision may opt-out by making a declaration within 100 days. But, given that no Party has been able over the past ten years to ignore the voluntary 1983/85 moratorium adopted by resolution [16], it is very unlikely that any Party could ignore the amendments, and dump radioactive wastes at sea. In any case, it is less likely that anyone would do so if the Convention formally bans the practice.

#### **1992 Paris Convention Compromise Formula**

At IGPRAD's Fifth Session last November, where the options were formally considered for the first time, a 7th option, based on the compromise agreed to in relation to the "Paris" Convention for the Protection of the Marine Environment of the NE Atlantic signed in September 1992 -- a permanent ban with an opt-out clause after the passing of a certain amount of time for some countries -- also was raised and included.

Negotiations for the new "Paris" Convention by the Parties to the Oslo and Paris Conventions began in 1989.[17] At an early stage, the Parties agreed that an Annex on "Dumping and Incineration at Sea" should implement the precautionary principle by establishing a "reversed listing of substances". Until recently, all international instruments regulating pollution were listing in their annexes substances that should not be dumped in the sea. By adopting a reverse approach, the new Convention would, for the first time, place the burden of proof on prospective dumpers to show why substances should be dumped. After it had been agreed that the new framework Convention's annex on dumping should ban all dumping activities with only a limited number of exceptions, radioactive wastes remained in brackets as a possible exception.

In the end, all the parties accepted an atypical compromise which banned the dumping of radioactive wastes at sea, but with an "opt-out" possibility after 15-25 years allowed for France and the UK only. Specifically, Article 3.3 of the Annex on the Prevention and Elimination of Pollution by Dumping or Incineration states that:

3.3(a) The dumping of low and intermediate level radioactive substances, including wastes, is prohibited.

3.3(b) As an exception to subparagraph 3(a) of this Article, those Contracting Parties, the United Kingdom and France, who wish to retain the option of an exception to subparagraph 3(a) in any case not before the expiry of a period of 15 years from 1st January 1993, shall report to the meeting of the Commission at Ministerial level in 1997 on the steps taken to explore alternative land-based options.

3.3(c) Unless, at or before the expiry of this period of 15 years, the Commission decides by a unanimous vote not to continue the exception provided in subparagraph 3(b), it shall take a decision pursuant to Article 13 of the Convention on the prolongation for a period of 10 years after 1st January 2008 of the prohibition, after which another meeting of the Commission at Ministerial level shall be held. Those Contracting Parties mentioned in subparagraph 3(b) of this Article still wishing to retain the option mentioned in subparagraph 3(b) shall report to the Commission meetings to be held at Ministerial level at two yearly intervals from 1999 onwards about the progress in establishing alternative land-based options and on the results of scientific studies which show that any potential dumping operations would not result in hazards to human health, harm to living resources or marine ecosystems, damage to amenities or interference with other legitimate uses of the sea.

Article 13 of the Convention, which is referred in Article 3.3(c) contains the rules by which decisions and recommendations will be reached:

Decisions and recommendations shall be adopted by unanimous vote of the Contracting Parties. Should unanimity not be attainable, and unless otherwise provided in the Convention, the Commission may nonetheless adopt decisions or recommendations by a three-quarters majority vote of the Contracting Parties.

The meaning of this albeit convoluted compromise is that -- at minimum -- ocean dumping of radioactive wastes is banned in the NE Atlantic until 2008, and that the ban will likely continue another 10 years, until 2018, by a three-fourth's majority vote.[18] As agreed, France and the UK have, for the first time, accepted formally to be bound by a ban on radioactive waste dumping. In addition, given that Article 3.2.(e) bans the dumping of vessels or aircraft from, at the latest, 31st December 2004, and that Article 3.3 ban radioactive waste dumping until at least 2008, the new Convention also represents a formal and definite abandonment of the ocean dumping option for decommissioned nuclear-powered vessels. [19]

## V. Subseabed Disposal of Radioactive Wastes

From 1973 onward, the OECD's Nuclear Energy Agency (NEA) acted as the Secretariat of a "Seabed Working Group" to coordinate research by their member States with a view to disposing of HLW under the seabed. This option, which consists of injecting canisters containing HLW under the seabed from a platform or a vessel, was considered in the NE Atlantic (Canary Islands and Madeira), the Caribbean, and the Pacific Ocean. The plans included shooting "suppository" shaped canisters intended to penetrate the ocean floor deep enough to prevent the release of radioactivity into the marine environment, and a drilling option, modelled on the engineering experience acquired in the context of the oil and gas offshore industry. From the mid-1970s to the mid-1980s, the US, France, the UK, Japan, Germany, and the Netherlands, among others, spent millions of dollars researching the technical and environmental feasibility of those options.

At the same time, those nations were careful to keep the issue outside of the LC for political reasons. The LC prohibits "disposal at sea" of HLW, but the question arose of whether disposal at sea referred to the final resting place of the wastes, or to the place where the disposal activity occurs. In 1983, Norway and Finland raised the issue for the first time, and the Parties agreed to convene a special legal experts meeting to consider whether subseabed disposal was covered by the LC. [20] However, no consensus was reached at that meeting, and the issue was forwarded to the 8th (1984) Consultative Meeting where, after a lengthy debate, two consensus points were adopted [21]:

1. The Consultative Meeting of the London Convention is the appropriate international forum to address the question of the disposal of HLW into the seabed, including the question of the compatibility of this type of disposal with the provisions of the Convention; [and]
2. No such disposal should take place unless and until it is proven to be technically feasible and environmentally acceptable, including a determination that such waste can be effectively isolated from the marine environment, and a regulatory mechanism is elaborated in accordance with the provisions of the London Convention to govern the disposal into the seabed of such radioactive wastes.

Beyond this basic agreement, two principal blocs expressed notably different views. The dominant coalition of nations believed that HLW disposal is covered by the LC and prohibited. They agreed that, while the treaty's express language may be unclear, protection of the marine environment required an interpretation that viewed subseabed disposal as "disposal at sea". In contrast, a small minority supported a US proposal taking the opposite view, namely that subseabed disposal was not covered by the LC, and therefore not prohibited. It was agreed that draft resolutions reflecting those two blocs would be attached to the report of the meeting, and tabled for subsequent consideration.



At the 11th (1988) Consultative Meeting the Irish delegation reopened the debate on subseabed disposal given the UK's plan to develop a program for intermediate- and low-level wastes in coastal areas into repositories accessed from the sea (via a platform) or from the shore (via a tunnel). At the 12th (1989) Consultative Meeting, Spain tabled a resolution stating that subseabed disposal of LLW into repositories accessed from the sea would be covered by the moratorium. At the 13th (1990) Consultative Meeting, that proposal was adopted by an overwhelming vote. [22] (See Table)

It is thought that, partly as a result of the debate within the LC, and the moratorium on the dumping of LLW at sea, as well as its extremely high cost, the subseabed disposal program has been virtually abandoned. However, periodically, in the US in particular, attempts have been made, especially in academic circles, to revive this option. As recently as 1992, for example, advocates of subseabed disposal for HLW were reported as being very active in Russia, possibly trying to establish a program there.

#### **VI. Amendment Conference: Toward the "Greening" of the London Convention**

Pursuant to discussions at their 14th and 15th Consultative Meetings, the LC Parties have agreed to hold in 1994 an Amendment Conference in the autumn of 1994 -- preceded by an "Amendments Negotiation Meeting" July 19-23, 1993, and the 16th (8-12 November 1993) Consultative Meeting, at which a decision is needed as to the nature and content of the amendments to be formally adopted at the 1994 Conference. [23]

Anticipating this process, the Danish delegation, supported by Norway and Iceland, tabled a proposal prior to the 15th Consultative Meeting containing a package of amendments addressing the controversies of the past decade. [24] The main features of the Danish proposal, which are likely to form the basis of discussions at the July 1993 negotiations and beyond, include: "the seabed and the subsoil thereof as part of the sea" in Article III(3) of the Convention, with a view to clarifying the status of the disposal of wastes in the subseabed; all radioactive wastes and other radioactive matter in Annex I's "black list"; "industrial wastes" in Annex I; a ban on incineration at sea of liquid noxious wastes; and the addition the precautionary approach to Annex III as the guiding principle of the Convention. As stated by the Danish delegation in their proposal [25]:

The substance and principles contained in most of the proposed amendments have already been agreed to in several Resolutions adopted by Consultative Meetings in the last decade.... Therefore, the proposal constitutes the much needed incorporation into the Convention of Decisions which already have been taken at past meetings.

For the London Convention, 1993-94 will be critical. In the past decade, the LC Parties have claimed that the Convention is "greening". It is perceived as moving away from providing a framework that regulates dumping at sea, and towards a more precautionary

and preventive approach. In that context, those countries firmly opposed to dumping at sea have increasingly assumed the mantle of LC leadership.

It also is true, though, that too few State Parties to the LC take a real interest and an active part in the work of that treaty regime. At best, of the 70 State Parties, no more than 45 ever show up at consultative meetings. And even among these, many fail to provide adequate reports on their dumping activities, despite legal obligations to do so. There remains the lingering perception that the lack of active involvement may be due to the fact that the LC is still a "dumpers' club". Under these circumstances, the recent decision to drop the word "dumping" from the name of the Convention was a step in the right direction. But more substantive reforms are now required. In this regard, the amendments proposed by Denmark, if adopted, would send a very positive message that the permissive era with regard to ocean dumping is finally over.

It also is important to note that ocean dumping was considered (until the moratorium) the most expeditious and least costly disposal method -- in political, social and economic terms. Accordingly, abandoning this option represents a significant incentive for the development and implementation of enlightened policies designed to reduce the generation of radioactive wastes at source, and to require governments to take full account of otherwise hidden costs of such wastes in present and future energy policies.

With the exception of illegal dumping activities of wastes from the USSR [26], officially no radioactive wastes have been dumped at sea since 1983. But the nuclear industry and the military are faced with ever-increasing quantities of wastes and an ever-growing opposition from citizens unhappy about the dumping of wastes in their own backyard. As a result, pressure to resume sea dumping has been maintained, and is expected to grow in the future unless the sea dumping option is definitively and permanently banned.

Beyond the important issue of ocean dumping itself, much more could be at stake: if the deliberate dumping of toxic industrial and radioactive wastes at sea can not be eliminated, there is little hope the international community will ever make significant progress toward eliminating marine pollution from toxic land-based sources -- roughly 80% of marine pollutant inputs worldwide. If the ocean dumping controversy -- limited in scope by comparison -- cannot be resolved after a decade-long review, it raises serious questions as to the international community's ability to effectively come to grips with the serious and growing problem of land-based sources of marine pollution, an issue which is far more complicated in technical, legal, political and economic terms.

**TABLE**  
**Votes on Radioactive Waste Dumping Issues at LC Consultative Meetings (1983-90)**

<b>1983 LLW Moratorium</b>	<b>1984 Legality of Subseabed Disposal</b>	<b>1985 LLW Moratorium</b>	<b>1990 Subseabed LLW</b>
<b>In Favor (19)</b>	<b>Nordic Resolution (25)</b>	<b>In Favor (26)</b>	<b>In Favor (29)</b>
Argentina	Argentina		Argentina
	Brazil	Australia	Australia
Canada	Canada	Brazil	Brazil
Chile	Chile	Canada	Canada
		Chile	Chile
			China
	Cuba	Cuba	C. d'Ivoire
			Cyprus
Denmark	Denmark	Denmark	Denmark
	Dominican Republic	Dominican Republic	
Finland	Finland	Finland	Finland
	Federal Republic of Germany	FRG	Germany
	Haiti	Haiti	
		Honduras	Honduras
Iceland	Iceland	Iceland	Iceland
Ireland	Ireland	Ireland	Ireland
			Italy
Kiribati	Kiribati	Kiribati	
			Malta
Mexico	Mexico	Mexico	Mexico
Morocco			Morocco
Nauru	Nauru	Nauru	Nauru
		Netherlands	Netherlands
New Zealand	New Zealand	New Zealand	New Zealand
Nigeria			Nigeria
Norway	Norway	Norway	Norway
		Oman	Oman
	Panama	Panama	
Papua New Guinea		Papua New Guinea	
Philippines		Philippines	
	Poland		
Portugal	Portugal		Portugal
			Solomon Island
Spain	Spain	Spain	South Africa
	St Lucia	St Lucia	Spain
Sweden	Sweden	Sweden	
	Yugoslavia		Sweden
	Zaire		
<b>Against (6)</b>	<b>US Resolution (6)</b>	<b>Against (5)</b>	<b>Against (4)</b>
Japan	France	France	France
Netherlands	Japan	South Africa	UK
South Africa	Netherlands	Switzerland	US
Switzerland	Switzerland	UK	USSR
UK	UK	US	
US	US		
<b>Abstaining (5)</b>		<b>Abstaining (7)</b>	<b>Abstaining (4)</b>
Brazil		Argentina	Belgium
FRG		Belgium	Greece
France		Greece	Japan
Greece		Italy	Switzerland
USSR		Japan	
		Portugal	
		USSR	

## FOOTNOTE REFERENCES

[1] Contracting Parties to the London Convention (at October 1992): Afghanistan - Antigua and Barbuda - Argentina - Australia - Belgium - Brazil - Belarus - Canada - Cape Verde - Chile - China - Costa Rica - Cote d'Ivoire - Croatia - Cuba - Cyprus - Denmark - Dominican Republic - Egypt - Finland - France - Gabon - Germany - Greece - Guatemala - Haiti - Honduras - Hungary - Iceland - Ireland - Italy - Japan - Jamaica - Jordan - Kenya - Kiribati - Libyan Arab Jamahiriya - Luxembourg - Malta - Mexico - Monaco - Morocco - Nauru - Netherlands - New Zealand - Nigeria - Norway - Oman - Panama - Papua New Guinea - Philippines - Poland - Portugal - Russian Federation - Seychelles - Solomon Islands - South Africa - Spain - Saint Lucia - Suriname - Sweden - Switzerland - Tunisia - Ukraine - United Arab Emirates - United Kingdom - United States - Vanuatu - (Yugoslavia) - Zaire.

[2] In the early 1980s, Japan and the US announced plans to initiate programs of radioactive waste dumping at sea. Japan was planning to dump up to 100,000 curies per year into a Pacific ocean site 600 miles North of the Northern Marianas. The US considered a plan to scuttle aging nuclear submarines in the Atlantic and Pacific oceans - as many as 100 old submarines, each representing 50,000 curies of radioactive wastes. In addition, the US Department of Energy wanted to dump at sea thousands of cubic meters of radioactively contaminated soils dating back from the early years of their nuclear weapons programme in 1940s, the so-called "Manhattan Project".

[3] In the summer of 1982, the largest dumping operation ever undertaken officially took place on the NEA NE Atlantic dump-site. Four ships were involved in dumping 10,000 tonnes of wastes from the UK, the Netherlands, Belgium and Switzerland, representing nearly 130,000 curies of radioactivity.

[4] Ocean dumping operations were coordinated by the OECD's NEA and were taking place in a designated dump site 700 km off the North West coast of Spain, in the NE Atlantic ocean, between 1967 and 1982. A total of 1,030,000 Curies were officially dumped on this site. Other dump sites in the NE Atlantic and the Gulf of Gascony were also used in the 1950s and 60s.

[5] Resolution LDC.14(7), "Disposal of Radioactive Wastes and Other Radioactive Matter at Sea", in Report of the 7th Consultative Meeting of the London Dumping Convention, Annex 3 (Document LDC 7/12).

[6] The International Atomic Energy Agency (IAEA) is the Vienna-based agency created in 1957 to "seek to accelerate and enlarge the contribution of atomic energy to peace, health and prosperity throughout the world" (Art II of the IAEA Statute), and the International Council of Scientific Union (ICSU) is a UN-based advisory scientific body.

[7] Resolution LDC.21(9), "Dumping of Radioactive Wastes at Sea", in Report of the 9th Consultative Meeting of the LC, Annex 4 (Document LDC 9/14).

[8] Resolution LDC.28(10) "Studies and Assessments pursuant to Resolution LDC.21(9)", in Report of the 10th Consultative Meeting of the London Dumping Convention, Annex 11 (Document LDC 10/15).

[9] Report of the 15th Consultative Meeting of the London Convention, Paragraph 11.2., Document LC 15/16, 3rd December 1992.

[10] "Options Paper on the Sea Disposal of Radioactive Wastes", Submitted to IGPRAD's 5th Session (Doc. LDC/IGPRAD 5/6);, and the Report of that Session.

[11] See, e.g., "Political Aspects of Radioactive Waste Disposal at Sea in the 1990s and Beyond", Greenpeace International (Document LDC/IGPRAD 5/2/1), and "Review of the Outcome of the UNCED", Greenpeace International (Document LDC 15/3/1).

[12] Resolution LDC.43(13) in Report of the 13th Consultative Meeting of the LDC, Annex 9.

[13] Document LDC/IGPRAD 5/6. Op. Cit.

[14] The decommissioning of nuclear installations, which are bound to becoming radioactive wastes themselves, may be portrayed as a special circumstance by the industry, although it is clear that it is not because it was foreseeable when decision to build those installations took place. Likewise, the recent implosion of the Soviet Union and future political and economic disorders in other regions could be seen as special circumstances, and in effect pervert the intent of the Parties to the Convention.

[15] The Paris Convention for the Protection of the N.E. Atlantic (1992), the Bamako Convention on the Ban of the Import into Africa and the Control of Transboundary Movement and Management of Hazardous Wastes within Africa (1991), the Barcelona Convention for the Protection of the Mediterranean (1973), the South Pacific Regional Environmental Programme (SPREP), among others.

[16] Although it points out deficiencies in the enforcement of the Convention, the case of the former USSR is a very special one.

[17] Belgium, Denmark, the EEC, France, Germany, Iceland, Ireland, the Netherlands, Norway, Portugal, Spain, Sweden, and the UK are members of the Paris Commission. All of them, plus Finland, are also members of the Oslo Commission.

[18] The UK accompanied its signature with the following statement: "The Government of the United Kingdom of Great Britain and Northern Ireland declares its understanding of the effect of the paragraph 3 of Article 3 of Annex II to the Convention to be among other things that, where the Commission takes a decision pursuant to Article 13 of the Convention, on the prolongation of the prohibition set out in subparagraph (3)(a), those Contracting Parties who wish to retain the option of the exception to that prohibition as provided for in subparagraph (3)(b) may retain that option, provided that they are not bound, under paragraph 2 of Article 13, by that decision." Article 13.2. states that:

A decision shall be binding on the expiry of a period of two hundred days after its adoption for those Contracting Parties that voted for it and have not within that period notified the Executive Secretary in writing that they are unable to accept the decision, provided that at the expiry of that period three-quarters of the Contracting Parties have either voted for the decision and not withdrawn their acceptance or notified the Executive Secretary in writing that they are able to accept the decision....

It is conceivable that the UK government may be technically right from a legalistic point of view. However, whether they are politically realistic remains to be seen, having been forced to renounce ocean dumping after a non-binding LC resolution in 1983.

[19] Ten British nuclear submarines are expected to be decommissioned in the UK by the year 2000, and France's oldest nuclear submarine has also been decommissioned recently. Until the adoption of the new Paris Convention, UK's Ministry of Defence was keeping ocean dumping as their preferred disposal option for these inconvenient bulky radioactive wastes.

[20] Report of the 7th Consultative Meeting of the LC, February 1983.

[21] Report of the 8th Consultative Meeting of the LC, 1984, page 31.

[22] Resolution LDC.41(13) in annex 7 of the Report of the 13th Consultative Meeting of the LC, October 1990.

[23] The 9th and 10th Consultative Meetings adopted resolutions on the "Procedures for the Circulation of Proposed Amendments of the [LDC]" and "Procedure for Preparation and Consideration of Amendments to Annexes to the [LDC]" respectively. The rule adopted suggests that amendments must be adopted "in principle" by a meeting that will designate a future meeting for the formal adoption of these amendments.

[24] In addition to radioactive wastes, the LC Parties have adopted non-binding resolutions phasing out incineration at sea of liquid noxious wastes (Resolution LDC.35(11)); phasing out (by 31 December 1995) the dumping of industrial wastes (Resolution LDC.43(13)); and on the precautionary approach (Resolution LDC.44(14)).

[25] "Proposal for a Draft Resolution on the Convening of a Conference in 1993 for Amending the [LC] and Proposals for Amendments to the Convention", Submitted by Denmark to the 15th Consultative Meeting of the LC, Document LDC 15/5/1.

[26] In 1991 and 1992, Greenpeace revealed that the USSR had been dumping secretly LLW and HLW (nuclear reactors with spent nuclear fuel) in the Kara and Barents Seas until at least 1986 (See "Necessary Correction to IAEA's Inventory of Radioactive Wastes in the Marine Environment: Soviet/Russian Dumping Activities", Greenpeace International (Document LDC 15/INF.18).



# GREENPEACE

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## **Radioactive Pollution from Accidents Involving Nuclear-Powered Vessels: Problems and Solutions**

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Research and Policy Priorities in the Arctic and North Atlantic,  
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### I. Introduction

The current concern over radioactive pollution of the Arctic and North Atlantic arises from a novel source: nuclear-powered vessels. Past worries have centered on radioactive releases from nuclear weapons testing (at Novaya Zemlya), nuclear weapons accidents (a B-52 crash at Thule, Greenland), deliberate dumping of nuclear waste (by Belgium, France, Germany, Italy, the Netherlands, Sweden, Switzerland, England and the United States), nuclear waste reprocessing plants in England and France (Sellafield and La Hague), and accidents involving civil nuclear plants (Chernobyl).

It is now evident that accidents besetting nuclear-powered vessels, particularly Soviet/Russian submarines, can have severe radiological consequences for the crews of the submarines and for the ocean environment. To provide information about this relatively unexamined possible source of radioactive contamination of the Arctic and North Atlantic, this paper will characterize the nuclear-fleets of the five nuclear-powers, examine some of the historical accident data for the Soviet/Russian, U.S., British, and Chinese nuclear navies, and discuss what is the likelihood of another serious accident. This paper will also argue that an important policy priority for the region under consideration would be ban on nuclear-propulsion at sea. Such a ban would be the ecologically optimal solution for avoiding radioactive pollution from accidents in the future. Due to the post-Cold War political, economic and military environment, such a ban is also more feasible than in the past.

### II. The nuclear-powered fleets of the U.S., Soviet/Russian, British, French and Chinese navies

The number of naval nuclear reactors is not generally appreciated. Although considerably smaller than their civil counterparts, naval nuclear reactors are more numerous.



Some 450 naval nuclear reactors were in operation in 1992, compared to 420 civil nuclear plants worldwide.<sup>1</sup>

The first nuclear-powered vessel, the submarine USS Nautilus was commissioned by the U.S. Navy in 1954. The Soviet Union soon followed, putting its first few nuclear-powered submarines to sea in the late 1950s. England launched its first nuclear-powered submarines in the 1960s, while France and China joined the nuclear-navy club with their own submarines in the 1970s.

Since 1954, some 480 nuclear-powered vessels have been commissioned. The vast majority have been constructed by the United States and Russia. By 1992, the Soviet Union/Russia had launched nearly 250 nuclear-powered vessels and the United States about 190. Britain, France and China collectively had built almost 40 nuclear submarines.

Of these nuclear-powered vessels, approximately 330 are general purpose submarines -- attack (SSN) or cruise-missile (SSGN). The next significant group consists of ballistic missile submarines (SSBN), some 120, carrying nuclear armed submarine launched ballistic missiles. Finally, the United States operates more than a dozen nuclear-powered aircraft carriers and cruisers, while Russia maintains about ten nuclear-powered cruisers and icebreakers along with one transport ship. There are several specialized nuclear-powered scientific and experimental submarines operated by the U.S. and Russian navies, too.

As for their locations, naval reactors are carried in all the world's oceans, with most situated, however, in the North Atlantic, North Pacific, and Arctic oceans, and the Mediterranean Sea. It is in these regions the nuclear-powered submarines and surface ships of the U.S., Russian, British and French navies operate, and have their naval nuclear bases and facilities.

Of particular interest is the concentration of activities and bases on the United States' East and West Coasts, England's Irish Sea and English Channel coasts, France's Atlantic and Mediterranean coasts, and Russia's European Arctic, Northern Pacific, and Sea of Japan coasts. Although submarines voyage throughout the deep oceans, operations of several countries vessels tend to converge near bases where intensive short-term domestic training deployments are complemented by long-term monitoring efforts by foreign submarines. As such the U.S. east and west coast, the British Irish Sea, and the Russian Arctic and North Pacific operating areas see relatively heavy nuclear-powered vessel traffic.

After thirty years of almost constant expansion, the future of nuclear-power of sea will be one of steady decline. The nuclear-powered fleets of the U.S. and Soviet navies

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<sup>1</sup> Greenpeace/Wise-Paris/Worldwatch Institute, The World Nuclear Industry Status Report: 1992, May 1992; Hans Kristensen, Naval Nuclear Propulsion After the Cold War: A Background Paper, (Revised), (Greenpeace, Washington, DC), September 1992.

peaked in the late 1980s. Decommissionings of vessels will greatly exceed commissionings in the next decade. By the year 2000, the number of nuclear-powered vessels at sea is projected to decline by over a half to just under 200 vessels. After the turn of the decade there will most likely be a slight further reduction to around 170 vessels. Beyond 2010, the size of the fleet becomes harder to predict as it will be dependent on building programs pursued in the latter half of this decade. The fate of these programs is a wide open question in the United States and Russia -- the two key countries -- and is dependent upon resolving fiscal and military-doctrinal crises in both countries. [See Figure 1]

The majority of nuclear-powered vessels are powered by pressurized-water reactors, ranging from 330-360 MW (th) to 48 MW (th).<sup>2</sup> All U.S. nuclear-powered submarines currently are powered by one nuclear reactor, while the majority of Russian submarines are powered by two (the U.S. built only one two-reactor submarine, the USS Triton). The U.K., French and Chinese submarines also utilize one reactor. As for surface ships, all but one are powered by two reactors. One U.S. aircraft carrier, the USS Enterprise, carries eight. Some 452 naval reactors carried by 298 vessels were counted operational as of mid-1992.<sup>3</sup>

The reactors' fuel enrichment varies from moderate to high depending on the country and class of vessel. U.S. naval reactor fuel is generally assumed to be enriched to greater than 93% U-235, whereas Russian reactor fuel seemingly can vary from 10 to 60% enrichment in most classes of submarines.

A few submarines have been powered by liquid metal cooled reactors. The U.S. built one submarine with a sodium-cooled reactor -- the USS Seawolf -- but the design was subsequently abandoned (the reactor vessel was removed from the submarine after a brief period of operation, defuelled, and dumped off the U.S. East coast in 1959). The Soviet Navy's use of liquid metal reactors has been more extensive, with lead-bismuth or even lead reportedly used as the coolant in a number of submarines. It is unclear whether sodium was ever employed.

The radioactive inventory of any given naval nuclear reactor can have a great variation depending on its design and length of operation. However, there is some official data concerning submarines which are known to have had accidents which is indicative of the radioactive inventories to be found in a naval reactor which could be lost at sea due to an accident.

In 1984, the U.S. Navy provided estimates of the radioactive inventory of the two submarines it has lost at sea: the USS Thresher in 1963 and the USS Scorpion in 1968. As of 1984, for each submarine, the Navy estimated that the corrosion products (crud) found in

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<sup>2</sup> Jane's Fighting Ships 1991-1992.

<sup>3</sup> Kristensen, p. 10.

the piping and reactor vessels was approximately 10 curies, mainly from Co-60 and Ni-63. The activation products in the reactor vessel, mainly Co-60 and Ni-63, contained less than 1,000 curies and 5,000 curies of radiation respectively for the Thresher and Scorpion. Finally, the Navy said the total fission and actinide series radioactivity in the nuclear fuel was less than 30,000 curies for each submarine.<sup>4</sup>

In March 1993, the Russian government furnished estimates of the radioactive inventories of the lost submarine reactors and warheads and the dumped reactors off Novaya Zemlya at the time of loss/dumping.<sup>5</sup> [See Table 1] As can be seen, the amounts of radiation are not inconsiderable. On average, lost reactors contained 130,000 Ci each, while dumped reactors had 330,000 Ci each.

### III. Discussion of Accident Data

#### A. Historical Accident Data

There is a wide variety of information available on accidents affecting nuclear-powered vessels. The vast majority relates to incidents other than those involving nuclear reactors, as this latter information is the most secret and closely held by the nuclear navies.

In the case of the U.S., the most comprehensive official source of information regarding accidents comes from the U.S. Safety Center released under the Freedom of Information Act. Data from 1980-1989, listing incidents on a case by case basis with location and some details about the accident, shows that U.S. nuclear-powered submarines experienced some 612 accidents. Ballistic missile submarines accounted for 26 percent and attack submarines 74 percent of this total.<sup>6</sup> To gain a fuller understanding of the safety

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<sup>4</sup> U.S. Navy, Annex to Appendix D, "Radiological Environmental Monitoring at Sites of Nuclear-Powered Submarine Thresher and Scorpion Sinkings," Final Environmental Impact Statement on the Disposal of Decommissioned, Defueled Naval Submarine Reactor Plants, May 1984, p. D-A6.

<sup>5</sup> Alexei Yablokov et al., Facts and Problems Related to the Dumping of Radioactive Waste in the Seas Surrounding the Territory of the Russian Federation (Materials from a government report on the dumping of radioactive waste, commissioned by the President of the Russian Federation, 24 October 1992 Decree no.613), Administration of the President of the Russian Federation Moscow, 1993, Tables 3 and 7. (hereafter, Yablokov Commission)

<sup>6</sup> Joshua Handler, Amy Wickenheiser, and William Arkin, Neptune Paper No. 4, Naval Safety 1989: The Year of the Accident, (Greenpeace, April 1990), p. 4. Additional information about naval accidents and trends is drawn from: Joshua Handler, "Submarine Safety -- The Soviet/Russian Record" Jane's Intelligence Review, July 1992; William Arkin and Joshua Handler, Neptune Paper No. 3, Naval Accidents 1945-1988 (Greenpeace/Institute for Policy Studies, June 1989).

record of the U.S. fleet, additional facts are available from news reports and ad hoc official documentation.

In the case of the other navies, no such official listing has yet been released, so historical data on accidents and accident trends is culled mainly from news reports, which at times is supplemented by sporadic official information.

Yet even without comprehensive and consistent data which would allow for comparison and statistical calculations, enough information exists to make some observations about the possibility of radiation entering the marine environment from an accident involving a naval nuclear reactor.

United States: the U.S. has lost two nuclear-powered submarines, the USS Thresher in 1963 off Cape Cod, and the USS Scorpion in 1968 in the mid-Atlantic. One nuclear reactor from the USS Seawolf was dumped early in the program after its performance was deemed to be unsatisfactory. Both submarines were lost in the 1960s and the reactor was dumped in 1959. There have been fires and collisions which reportedly could have led to a vessel sinking (USS Enterprise, 1969, fire and explosions; USS Tautog, 1970, collision with Soviet submarine).

Since the 1960s, the U.S. Navy has not experienced any disastrous loss, but there have been some close calls. A regular series of collisions, fires, and floodings beset U.S. nuclear-powered submarines like any vessel. Most recently was the collision with a Russian Delta SSBN off the Kola peninsula in March 1993. According to Russian Navy calculations, had the collision occurred several seconds later, the U.S. submarine would have hit the Russian submarine solidly amidships rather than just glancing off its bow.<sup>7</sup> Another collision between a Russian and U.S. submarine in February 1992 in the same area caused enough damage that the U.S. submarine -- the USS Baton Rouge -- is now up for retirement.

Thus, however seemingly remote, the loss of a U.S. nuclear-powered vessel due to external events, particularly like collisions with other vessels, remains an ever present concern. In this regard it is useful to quote at length from the U.S. Navy Safety Center's survey of selected ship collisions:<sup>8</sup>

In the past 10 years there have been five cases of submarines, at periscope depth, colliding with surface ships. The history of submarine disasters yields chapter after chapter of collision at periscope depth, or en route to periscope depth. The first was the S-4 in 1927 .... Hit by a surface ship from the beam, a submarine will crack like

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<sup>7</sup> Aleksander Mozgovoy, "Twenty Meters Away from Nuclear Catastrophe," Rossiyskaya Gazeta, 1 April 1993 (translated in FBIS-SOV-93-061, p. 5).

<sup>8</sup> U.S. Navy Safety Center, Survey 2 of Selected Ship Collisions, n.d., p. 30, released under the Freedom of Information Act.

an eggshell. This was demonstrated by STICKLEBACK in the late 50's. Numerous nuclear submarine collisions have caused severe damaged to the sail. It is only a coincidental matter of relative depths that have saved us from further catastrophes. [emphasis added]

Soviet Union/Russia: The Soviet Navy has suffered a series of severe nuclear submarine accidents. Three nuclear submarines have sunk and not been recovered (a November SSN in 1970 off Spain, a Yankee SSBN in 1986 off Bermuda, and the Komsomolets SSN in 1989 off Norway). Another sank but was raised, a Charlie SSGN in 1983 off Petropavlovsk. In addition there have been a series of severe reactor accidents which permanently or temporarily incapacitated a submarine. Subsequently, reactors from some of these damaged submarines were dumped at sea. [See Tables 2 and 3]

#### B. Likelihood of future serious accidents

What is most worrisome for the future is the consistency of the Soviet Navy's safety record. Whereas conceivably an improvement overtime could be postulated as the Soviet Navy climbed up the learning curve, the opposite almost seemingly has transpired. Almost 30 years into the program, remarkably, two nuclear submarine reactors had explosions and meltdowns within the space of four months in the Soviet Pacific Fleet in 1985, and two nuclear submarines sank at sea between 1986 and 1989.

The Soviet nuclear submarine force never had a golden age. Terrible disasters occurred under relatively optimal societal conditions. The current situation in Russia and the fleet, even if less submarines are operating and with remedial safety programs underway, does not augur well for the future. The likelihood of a severe accident at sea with the possible loss of a submarine or radiation leakage in the next 5-10 years due to internal causes (fires, explosions, floodings, human error, etc.) is relatively high. Added to this is the possibility of events for which the Russian Navy cannot be wholly responsible for, like collisions with foreign submarines or surface vessels.

England, France and China: The situation in England and France parallels that of the United States but on a much smaller scale, due to the smaller size of their forces. Neither country has lost a nuclear-powered submarine, but both have suffered naval disasters. In the absence of better information, the concern about both fleets is the same as the United States: nuclear-powered submarines are not inherently any safer or unsafer than any other warship. In the case of China, reliable information is available about its half-dozen nuclear-powered submarines, although one 1991 account describes the submarines to be in very poor

condition.<sup>9</sup>

Thus in regards to two main avenues for naval nuclear reactors to be permanently deposited in the oceans due to accidents -- deliberate dumping of damaged reactors or accidental loss -- the chances of either of occurring should be considered as the following:<sup>10</sup>

**Deliberate dumping of damaged submarines:** although this remains a possibility, particularly for the three wrecked submarines in the Russian Pacific Fleet, the probability of this occurring is low; but mainly due to political developments. The recent political attention focussed on the past dumping of damaged naval nuclear reactors will remain a deterrent to the deliberate dumping of nuclear reactors. Changing the London Convention's moratorium on nuclear dumping into a permanent ban would create more political pressure not to dump damaged nuclear submarines at sea.

**Accidental loss of nuclear-powered vessels:** compare to deliberate dumping, the probability of the sinking of a nuclear-powered vessel due to internal or external causes is much higher. The causes of past nuclear submarine sinking are typical of the major problems facing warships at sea: equipment malfunctions (the Thresher); accidents involving ordnance and missiles (the Scorpion and Yankee); fires (November and Mike) and floodings (the Charlie). Since these hazards are inseparable from the problems of operating large vessels with complicated technologies in a demanding environment, which time and again in surface ships and submarines has led to disasters, it is only a matter of time before another serious accident occurs which takes a nuclear reactor to the bottom.

### C. Accident Consequences: Can Releases of Radiation Occur

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<sup>9</sup> Jack Anderson and Dale van Atta, "Chinese Navy Far From Shipshape," Washington Post, 5 December 1991.

<sup>10</sup> Two other avenues, however, need to be mentioned even though they are not within the scope of this paper: dumping of decommissioned submarines and the loss of nuclear-powered vessels in wartime.

Until the London Convention's moratorium on nuclear waste dumping becomes a permanent ban, England and Russia, and possibly France and China, will have the option to be able to dispose of their decommissioned submarines at sea. (The United States under current plans has no intention to do so.)

Nuclear-powered ships have operated in hostilities -- e.g. Vietnam, the Falklands, the Iraq war -- so far without any misadventure. But with the proliferation of mine technology, anti-ship missiles, and submarines among non-nuclear powers, the chances are increasing that in the next conflict or even during peacetime patrols a nuclear-powered submarine or surface vessel will be successfully attacked and damaged, raising the prospect of serious incident involving a nuclear reactor.

The loss or dumping of a nuclear reactor (or nuclear weapon) although creating the possibility, does not in and of itself spell immediate disaster for the marine environment or human populations dependent on it. There are numerous factors determining whether radiation will be released, transported, and enter the human food chain: the status of the emergency reactor shutdown or cooling systems; the fuel's integrity; the condition of the reactor vessel or primary coolant piping; the state of the reactor compartment and vessel hull; which radionuclides are present; what oceanographic conditions prevail in the area of the accident; what biological activity exists in the vicinity of the wreck, etc.

The U.S. and Russian navies are particularly reassuring on these points. The U.S. Navy claims that, "The reactor's many protective devices and inherent self-regulating features are designed to prevent any melting of the fuel elements." Also, the Navy notes the most dangerous radionuclides in the spent fuel will be immobilized since the fuel, "will remain intact for an indefinite period of time." Several Navy surveys near the sunken Thresher and Scorpion submarines have detected only trace amounts of activation products indicating some primary coolant leakage, but no fuel element failures.<sup>11</sup>

When the Yankee and Mike submarines sank, Russian authorities claimed the reactors were shut down precluding a radiation emergency. In regards to the Yankee in 1986, the Soviet government announced, "the immediate cause [of the sinking] is the speedy flooding of water from the outside. The reactor has been shut down. According to the conclusion of specialists, the possibility of a nuclear explosion and radioactive contamination of the environment is excluded."<sup>12</sup>

In the case of the Mike in 1989, Soviet Defense Minister Dimtri Yazov said, "We know for sure that the nuclear reactor is shut down. In the opinion of competent services, a radioactive contamination of the environment is ruled out."<sup>13</sup> Ministry of Foreign Affairs spokesman Gennadiy Gerasimov said, "the nuclear power installation was reliably shut down and is now cooling and there is no danger of radioactive pollution."<sup>14</sup> According to information provided by the Soviet Northern Fleet, "To ensure radiation safety, the nuclear-powered engine was stopped and the power unit was effectively blanked off. Destruction of the reactor vessel is ruled out."<sup>15</sup>

<sup>11</sup> U.S. Navy, Environmental Impact Statement, Annex to Appendix D, pp. D-A1 ff.

<sup>12</sup> George Wilson and R. Jeffrey Smith, "Crippled Soviet Sub Sinks in Atlantic; No Nuclear Contamination Expected," Washington Post, 7 October 1986.

<sup>13</sup> TASS, 8 April 1989 (in FBIS-SOV-89-067, 10 April 1989, p. 6).

<sup>14</sup> TASS, 10 April 1989 (translated in FBIS-SOV-89-068, 11 April 1989, p. 4).

<sup>15</sup> Vasily Belousov, TASS, 9 April 1989 (translated in FBIS-SOV-89-067, 10 April 1989, p. 7).

Interestingly, in light of today's controversy over the radiological threat posed by the sunken Mike's nuclear weapons, the same sources said, "the design of the warheads completely rules out radiation threat during very deep submergence."

As for the dumped submarine reactors, when the secret dumping was finally revealed in February 1992, Russian Admiral Vitaly Zaitsev was quoted as saying, "reactor sections of the worked-out submarines are also stored in offshore waters. The sections have been made thoroughly hermetic and are absolutely non-radioactive from the outside."<sup>16</sup>

Thus the navies portray a reassuring picture. If an accident occurs, emergency systems guarantee the shutdown and safety of the reactor. When a reactor sinks its design precludes any release of fission products or actinides. If dumped, the reactors are sealed sufficiently to isolate them from the marine environment. (And, in the case of lost nuclear weapons, no danger exists.)

In the case of the United States, secrecy precludes an independent evaluation of the Navy's claim. Although the data gathered around the Thresher and Scorpion sites is taken to be valid, it is necessary to have more information about the state of the submarines after their accidents, alloys in the reactor vessels, their fuel, time of operation, etc., before any independent conclusions about prospective leaks and dangers can be reached.

In the case of the Soviet Union/Russia, new information has come to the fore in 1991-1992 which cast doubts about its reassurances and raises concerns for a future accident.

In terms of emergency systems to protect the reactor, they do not always function as planned: when the Yankee submarine was stricken by explosions and flooding, the emergency system only shut down one reactor. A malfunction kept it from shutting down the second. As a result, sailors had to heroically enter the reactor compartment and manually lower the control rods. One became trapped in the compartment and died.<sup>17</sup> Indeed in the case of submarine collisions, according to one Russian admiral, there may not be time to shut down the reactor, meaning a nuclear disaster could ensue.<sup>18</sup>

In regards to deliberately dumped materials: the Yablokov Commission report contains evidence that some problems may have already occurred:<sup>19</sup>

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<sup>16</sup> Sergei Shargorodsky, "Soviet-Sea Dumping," Associated Press (Moscow), 28 February 1992.

<sup>17</sup> Alexander Mozgovy, "The Agony of the Atomic Leviathan," Golos, no. 17, April 27 - May 3, 1992; Sergei Shargorodsky, "Submarine Disaster," Associated Press (Moscow), 21 January 1993.

<sup>18</sup> Joshua Handler, Trip Report: Greenpeace Visit to Moscow and Russian Far East July - November 1992 Subject: Russian Navy Nuclear Submarine Safety, Construction, Defense Conversion, Decommissioning, and Nuclear Waste Disposal Problems, 15 February 1993, p. 6.

<sup>19</sup> Yablokov Commission, Section 3.3.

In regards to sunken weapons, the current controversy over the status of the sunken Mike nuclear warheads is indicative that the Soviet authorities guarantees were premature.



It must be kept in mind, however, the possibility of accidental contact with individual quite dangerous items in the future, as result, for example, of removal during diving work, storm throw-out onto the shore from shallow bays of the islands of the Novaya Zemlia archipelago, which hold a significant portion of the dumped RAW.

In order to avoid accidental human contact with radiation contaminated objects thrown out onto the shore, the Northern test site Novaya Zemlia conducts annual visual examination of the coastal zone of the eastern coast of the islands of the Novaya Zemlia archipelago.... No items or artifacts of radiational danger were discovered, with the exception of 1984, when on the coast of the Abrosimova Bay on the Novaya Zemlia archipelago an identified metal structure was discovered emitting high level of radiation (more then 100 R/hour) -- fragments of fuel element. [emphasis added]

Thus, although no substantial release of radiation has been detected near sunken U.S. submarines, the question is still in doubt for dumped and lost Russian nuclear submarines. And, the situation may be different in the future, either in the form of leaks from currently dumped materials, or from an accident where emergency systems fail to protect the reactor's fuel from damage.

#### IV. Conclusion: Banning nuclear-propulsion at sea

The reduction in size and operations of the nuclear navies' nuclear-powered fleets could lead to a reduction in the number of accidents. However this is not assured. Both the U.S. and Russian nuclear-submarine fleets experienced some of their worst disasters when the number of nuclear submarines was low. Also, the two recent collisions between U.S. and Russian nuclear-powered submarines indicate that even lower operational tempos do not mean necessarily less serious accidents.

Although there are several lesser steps which could be taken to reduce the possibility of a serious accident -- improved internal safety measures, better rules of the road governing nuclear submarine operations, limits on nuclear submarine operations -- and also steps which could be taken to improve accident response -- better damage control, quick notification of problems, assembling international rescue and remediation teams -- the optimal solution for eliminating the naval nuclear danger to the marine environment is a ban on nuclear propulsion at sea.

Several military, financial, and political factors make this solution both desirable and feasible.

Nuclear-powered attack submarines were uniquely suited to the Cold War and a conflict between NATO and the Warsaw pact. With the end of the Cold War and U.S.-Soviet political confrontation, the U.S. nuclear submarine force lacks any significant mission. Since the majority of Soviet nuclear-powered general purpose submarines were for attacking

U.S. submarines and carrier battle groups, their missions, too, are marginal. Already the number of general purpose submarines are declining precipitously and building programs have been greatly reduced.

Nuclear-powered ballistic missile submarines are also in decline. The U.S. force will dwindle from a highpoint of some 40 submarines in the 1970s to 18 by the end of the decade. Conceivably, these numbers may be further slightly reduced. In any event, the first eight Ohio class Trident submarines will begin to be retired around 2011 at the end of their 30 year life-span. Currently, no new ballistic missile submarines are under design. The Russian force is also being reduced dramatically. Approximately 40-50 ballistic missile submarines will be retired as the Russian force shrinks to some 20 submarines by the end of the decade. Again, this force could also decline further.

The START II treaty seemingly puts a premium on warheads carried by nuclear-powered ballistic missile submarines (1,750 warheads, some half of the 3,000 - 3,500 allowed by either side). However, as the U.S. and Russian militaries (as well as the French and British) continue to reorient their strategies towards third-world conflicts, the fleet of ballistic missiles carried on submarines will look increasingly useless compared to tactical nuclear weapons in the military's eyes.<sup>20</sup> Submarine warheads will also be increasing expensive to maintain on station in comparison to land-based missiles and aircraft carried bombs and missiles. Another indication that this trend may be developing is that Russia has reduced its ballistic missile patrols. According to the U.S. Navy's Chief of Naval Intelligence, "[T]he average number of Russian SSBNs at sea on patrol at any given time declined to approximately one third of 1991 levels."<sup>21</sup>

New nuclear-powered vessels are becoming increasingly expensive even as the decommissioning costs for older vessels are becoming apparent. The new U.S. Seawolf nuclear submarines will cost more than a \$1 billion each. Their anticipated follow-on, the Centurion class will likely cost almost as much per unit. Decommissioning costs for 100 U.S. submarines, including the scrapping of 85, are estimated to be \$2.7 billion by the end

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<sup>20</sup> Already there is some evidence that tactical nuclear weapons may be brought under strategic control and planning, which means they will begin to compete with more traditional strategic weapons for roles and missions. Although he defends the Trident submarine ballistic missile system, General Lee Butler, Commander-in-Chief US STRATCOM noted in his testimony to the Senate Armed Services Committee that "At the behest of the Chairman of the Joint Chiefs of Staff, Strategic Command is working with selected regional Unified Commands to explore the transfer of planning responsibilities for employment of nuclear weapons in theater conflicts;" General Lee Butler, USAF, Commander-in-Chief US STRATCOM, Testimony before the Senate Armed Services Committee, 22 April 1993, p. 3.

<sup>21</sup> Director of U.S. Naval Intelligence, Rear Admiral Edward Shaefer, "Annual Posture Statement," 12 May 1993, p. 46. This means may be only one or two SSBNs may be on patrol.

of the decade.<sup>22</sup> The crisis facing the Russian Navy in decommissioning its nuclear-powered submarines is well described by serving Russian naval officers<sup>23</sup> and in Section 4 of the Yablokov Commission report.

Port calls by nuclear-powered vessels have been controversial since the 1960s. Several countries — Iceland, Denmark, New Zealand — ban visits, and in other countries, like Japan, their visits have been a matter of controversy. Also, twice in the late 1980s, Iceland brought up the subject of a safety regime for nuclear-powered vessels in the North Atlantic at the United Nations in the aftermath of the Mike submarine sinking. If another serious accident befalls a nuclear vessel or if one were damaged in wartime, more political opposition to their presence may arise.

In short, in the cost of nuclear-power at sea is increasingly outweighing any putative military benefits. The threat nuclear naval vessels pose to non-nuclear and nuclear countries alike even in peacetime demands that steps are taken or an agreement is reached soon on a global ban on nuclear propulsion. A ban would not eliminate all sources of radioactive threats to the ocean environment, but it would abolish a radioactive threat that is particularly pernicious due to the secretiveness and widespread nature of nuclear-submarine operations, an naval nuclear accident's ability to affect innocent bystanders — the oceans and non-nuclear countries — and submarine operations' frequent proximity to rich fishing areas.

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<sup>22</sup> U.S. General Accounting Office, Nuclear Submarines: Navy Efforts to Reduce Activation Costs, NSIAD-92-134, July 1992, p. 2.

<sup>23</sup> See: Joshua Handler, "No Sleep in the Deep for Russian Subs," Bulletin of the Atomic Scientists, April 1993, pp. 7-9; Joshua Handler, Trip Report: Greenpeace Visit to Moscow and Russian Far East...

Table 1: Radioactive Inventories of Soviet  
Nuclear Reactors and Weapons Lost or Dumped at Sea

	Area, Year	Total Activity (max) kCi
Submarine with 2 reactors, 1 with spent nuclear fuel	Abrosimova Bay, 1965	800
Section with 2 reactors with spent nuclear fuel from submarine	Abrosimova Bay, 1965	400
Screen assembly from icebreaker Lenin with remaining spent nuclear fuel	Syvolky Bay, 1967	100
Reactor from a submarine with spent nuclear fuel	Novozemelskay a Depression, 1972	800
Submarine with 2 reactors with spent nuclear fuel	Stepovogo Bay, 1981	200
<u>sub-total dumpings</u>		<u>2,300</u>
November SSN (two reactors and two nuclear weapons)	Bay of Biscay, 1970	250
Yankee SSBN (two reactors and 34 nuclear weapons)	Bermuda, 1986	250
Komsomolets (one reactor and two nuclear weapons)	Norwegian Sea, 1989	159
<u>sub-total losses</u>		<u>659</u>
<u>Total (reactor inventories only, weapons add 6.03 kCi)</u>		<u>2,959</u>

**Table 2: SERIOUS SOVIET/RUSSIAN NUCLEAR-POWERED SUBMARINE  
ACCIDENTS 1960-1990**

1960s

- JUL 61 K-19 HOTEL SSBN; SERIOUS LOSS OF COOLANT ACCIDENT IN NORTH ATLANTIC. REACTOR(S) SUBSEQUENTLY DUMPED OFF NOVAYA ZEMLYA.
- MAY 68 K-27 LIQUID-METAL NUCLEAR-POWERED SUBMARINE IN NORTH ATLANTIC. COOLANT FREEZE CAUSES MAJOR RADIOLOGICAL ACCIDENT. SUBMARINE WITH REACTORS AND NUCLEAR FUEL DUMPED OFF NOVAYA ZEMLYA.
- MAR 68 K-129 GOLF SSB SINKS 8-10 MARCH ABOUT 700 MILES NNW OF MIDWAY ISLAND  
1970s WITH FIVE NUCLEAR WEAPONS. PARTS LATER RECOVERED BY THE CIA IN 1974.
- APR 70 K-8 NOVEMBER SSN; FIRE LEADS TO SUB SINKING. TWO NUCLEAR REACTORS AND TWO NUCLEAR TORPEDOES SINK WITH SUB IN THE BAY OF BISCAY. SUBMARINE UNRECOVERED.
- 1972 ALFA SSN IS DISMANTLED AFTER SUFFERING AN ACCIDENT INVOLVING THE REACTOR IN NORTH ATLANTIC. FATE OF REACTOR UNKNOWN.
- JUL 79 K-116 (ORDER NO. 541) ECHO I SSN; ONE REACTOR SUFFERS A MELTDOWN WHILE AT SEA NEAR VLADIVOSTOK. SUBMARINE REMOVED FROM SERVICE. STORED  
1980s AT THE PAVLOVSK NUCLEAR SUBMARINE BASE NEAR VLADIVOSTOK.
- APR 82 K-123 ALPHA SSN; PRIMARY CIRCUIT DESTROYED WHILE IN NORTH ATLANTIC. REPAIRS LASTED 9 YEARS. FATE OF REACTOR UNKNOWN.
- JUN 83 K-429 CHARLIE SSGN; SINKS OFF PETROPAVLOVSK. RAISED BUT SINKS AGAIN AT DOCKSIDE. REMOVED FROM SERVICE.
- AUG 85 K-431 (ORDER NO. 175) ECHO II SSGN; ONE REACTOR EXPLODES DURING REFUELLING AT THE CHAZHMA BAY FACILITY NEAR VLADIVOSTOK. SUBMARINE REMOVED FROM SERVICE. STORED AT THE PAVLOVSK NUCLEAR SUBMARINE BASE NEAR VLADIVOSTOK.
- DEC 85 K-314 (ORDER NO. 610) CHARLIE SSGN; SUFFERS A MELTDOWN IN ITS REACTOR DUE TO EQUIPMENT FAILURE IN THE PRIMARY CIRCUIT AS THE SUB WAS RETURNING TO BASE NEAR VLADIVOSTOK. CURRENTLY AT PAVLOVSK.
- OCT 86 K-219 YANKEE SSBN; SUFFERS EXPLOSION IN MISSILE TUBE EAST OF BERMUDA AND SINKS. TWO NUCLEAR REACTORS AND 34 NUCLEAR WARHEADS SINK WITH SUBMARINE. SUBMARINE IS UNRECOVERED.
- APR 89 K-278 KOMSOMOLETS MIKE SSN; FIRE ON BOARD LEADS TO SINKING SOUTH OF BEAR ISLAND. TWO NUCLEAR TORPEDOES AND ONE NUCLEAR REACTOR SINK WITH SUBMARINE. SUBMARINE IS UNRECOVERED.

TOTAL 12

Table 3: TOTAL NAVAL NUCLEAR REACTOR VESSELS, REACTOR SCREENS,  
SUBMARINE NUCLEAR WEAPONS  
DUMPED OR LOST IN THE WORLD'S OCEANS

### NUCLEAR REACTORS

#### SOVIET UNION/RUSSIA

- 18 REACTORS DELIBERATELY DUMPED AS A RESULT OF ACCIDENTS. 16 IN ARCTIC AND 2 IN PACIFIC. SIX CONTAINED THEIR FUEL. (AT LEAST INVOLVING SUBMARINES K-19, K-27, K-11, K-140, K-3, K-5, K-22 AND ICEBREAKER LENIN.)
- 2 SCREEN ASSEMBLIES FROM NAVAL REACTORS. ONE CONTAINED FUEL IN THE ARCTIC. (LENIN AND UNKNOWN SUBMARINE).
- 5 LOST AT SEA DUE TO ACCIDENTS (NOVEMBER 1970 (2); YANKEE 1986 (2); MIKE 1989 (1))

#### UNITED STATES

- 1 REACTOR DELIBERATELY DUMPED IN 1959 FROM SEAWOLF SUBMARINE. DEFUELLED, OFF ATLANTIC COAST.
- 2 REACTORS LOST AT SEA DUE TO ACCIDENTS (THRESHER 1963 (1); SCORPION 1968 (1)).
- 28 TOTAL REACTORS AND SCREENS FROM NUCLEAR-POWERED NAVAL VESSELS DUMPED OR LOST AT SEA.

### NUCLEAR WEAPONS

#### SOVIET UNION/RUSSIA

- 43 5 - GOLF 1968; 2 - NOVEMBER 1970; 34 - YANKEE 1986; 2 - MIKE 1989.

#### UNITED STATES

- 2 2 - SCORPION 1968.
- 45 TOTAL NUCLEAR WEAPONS FROM SUBMARINES

Est. Commissioned Nuc-Powered Vessels in  
US, Soviet/Russian, UK,  
French, Chinese navies

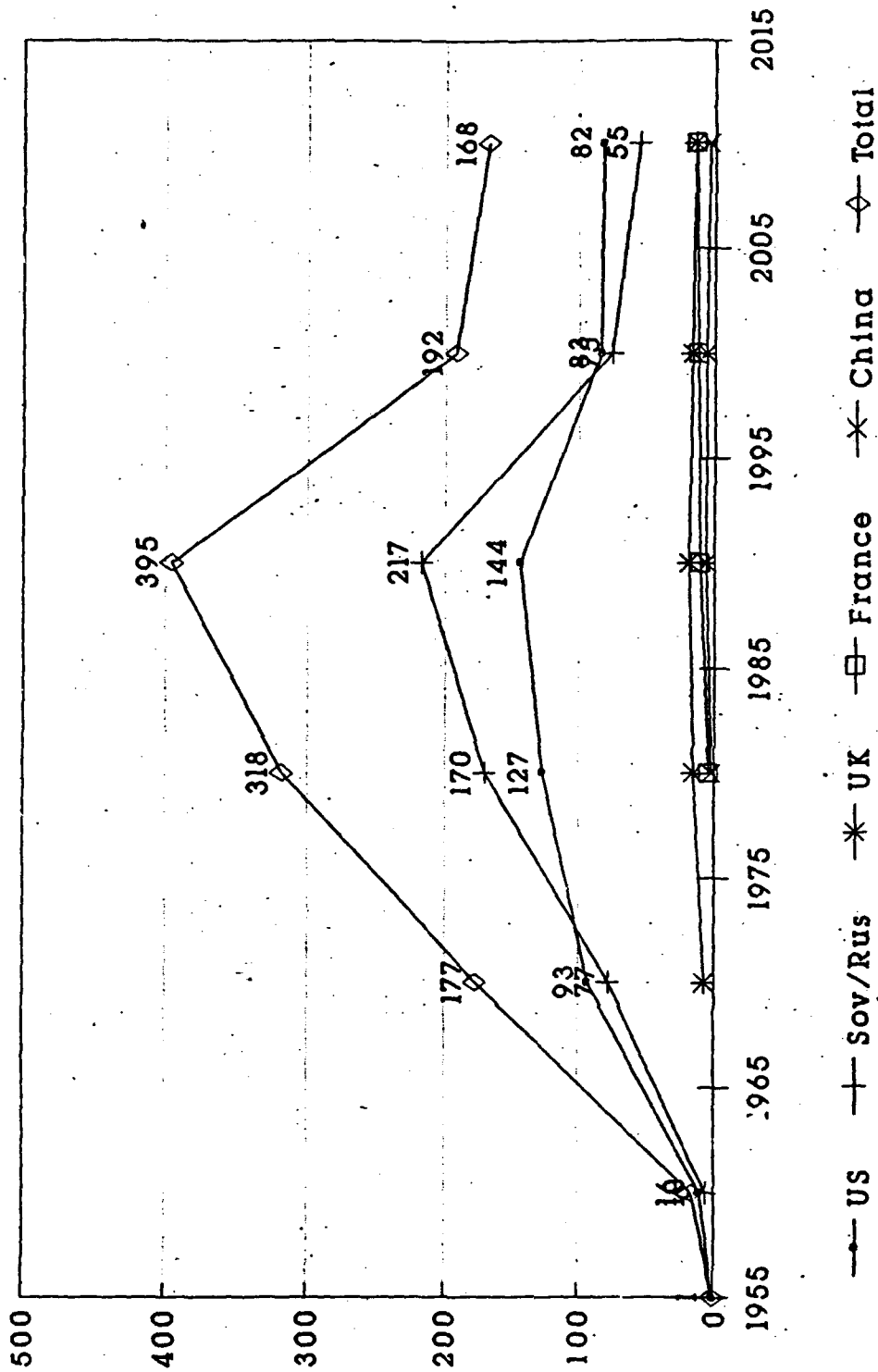


Fig 1. Handler/Greenpeace, June 1993

Use of ALARA in Decision Making on Costly  
International Cleanup Effort of Radioactive Contamination

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1. INTRODUCTION.

In this paper we consider the problem of transboundary risk management. We discuss a situation when an external source of hazard contributes heavily to the total public and environmental risk in a country.

A particular case of external military threat was widely discussed in the seventies. The major efforts had been made to reduce military risks and were resulted in a concept of the strategic parity. This concept formulated basis for a compromise decision making. It is important to note that compromise decisions are corporate by nature. It means that they regulate corporate (cooperative) actions of independent states aiming at a certain common goal (in this case international strategic stability).

Now the world community faces another acute problem - mutual environmental threat. And, correspondingly, the development of environmental parity concept is urgently needed. The authors propose a very general model of societal risk management and on the basis of this framework show that for the problem of transboundary risk management a mutually acceptable cooperative solution can be found. The solutions of this kind may serve as a foundation for the environmental parity concept.



It should be stressed out that the existing experience in settling down political and economic conflicts between countries is insufficient if we switch on to ecological problems. Therefore a specially developed approach and theory is essential for tackling with environmental conflicts. Here we make an attempt to develop the framework for settling down such conflicts.

The paper is outlined as follows. Section 2 presents the basic concepts of the framework to investigate the problem of risk management. Section 3 proposes a solution for the problem of optimal investments into both the internal and external source of hazard. Possible implications of the proposed model are discussed in Section 4.

## **2. CONCEPTS OF THE FRAMEWORK.**

The development of the methodology for optimal risk management in Russia was started in the beginning of the eighties in Kurchatov Institute by V. Legasov and I. Kuz'min. These scholars proposed the following definition: *Safety is the state of being protected from undue hazard.* Thus before any further consideration several important questions must be answered: Who is "Being protected" - an individual, society or its environment? What is excessive and what is palatable? And why people are dying? Here we shall focus on these questions.

### **2.1 SAFETY FOR THE GROUP.**

We shall restrict ourselves to setting a safety policy for the group. In fact this group will be the population of a country or even of several neighbouring countries. Thus we do not consider personal risk and decision making. These matters are substantially discussed by R. Keeney in [1,2]. Neither do we consider risks to the environment. One can find a good explanation why and when it is reasonable to examine risks to the environment as an independent task of risk management in [3,10].

In the case of setting a safety policy for the group the statistical nature of risk management plays an important role because the individual gainers and losers can not be determined *a priori*. In the proposed approach life expectancy is a measure of human safety. An interesting discussion why life expectancy provides an effective indicator of the level and potential improvements in human safety may be found in [4].

## 2.2 BIOLOGICAL OR SPECIES LIFE LONGEVITY.

*In this lifetime feel fear of god.*  
Warner Bros. Inc.

How much is the maximum achievable life expectancy? Why can not man live forever? We may recommend a book by L. Gavrilov which contains a review of existing theories of resources of human organism [5]. L. Gavrilov estimates the biological life longevity  $T_{max}$  of *homo sapiens* as 94.5 years. It implies that an average human every year experiences the risk of "natural death"  $r(t)$ , where  $t$  is age in years. This risk is the probability of dying at age  $t$  due to the limited resources of human organism. It can not be managed or reduced. Given  $r(t)$  one can easily calculate  $T_{max}$ . The inversed value  $T_{max}^{-1}$  we may define as mean biological death rate  $r$ .

## 2.3 MANAGEABLE RISKS.

To achieve this maximum - biological life longevity - seems to be highly improbable even in the distant future because great many additional risks which we call manageable risks are presented in the society. These risks are of different origin. On early stages of the civilization they were mostly natural hazards resulting from unfriendly environment. In order to reduce these natural dangers socio - economic sphere had been built. It in turn brought out technogenic and socio - economic risks. Despite the very complex character of socio - economic evolution of a given society (or of the human civilization as a whole) one fact remains clear: Life

expectancy gradually increases over time. However instead of time analysis we shall provide here "spatial analysis" i.e. compare different countries at present.

Now suppose that during the year  $j$  an average human in the population experiences an additional risk  $R(j)$ . One can calculate the resulting reduction of life expectancy  $\Delta T$ .

$$\Delta T = R(j) * T_{max} * (1-r)^{j-1}$$

This expression was derived assuming that  $r(t) = \text{Const} = r$  and henceforth pursues only illustrative purpose. What is important though is that the amount of life lost is proportional to the additional risk  $R$ . The last factor  $(1-r)$  shows that the later risk  $R(j)$  appears the less it affects life expectancy. If we knew all additional risks which threaten average human life from birth to death (given age - adjusted death rate) we could have calculate life expectancy  $T$ :

$$T = T_{max} (1 - \sum_{j=1} R(j) * (1-r)^{j-1})$$

It is these risks that we want to manage. Again, the goal of safety objective is maximization of life expectancy for the entire group affected. This is of course only a zero approximation of safety objective for it does not take into account life quality. See the definition of quality adjusted life expectancy in [5].

So how can we manage risks?

## 2.4 RICHER IS SAFER

*The cost of a thing is the amount of life which is required to be exchanged for it, immediately or in the long run.*

Henry David Thoreau

Several quantitative studies have investigated the relationship between the level of socio - economic development

of a country and mortality rates [6,7]. Although large uncertainties always present in this kind of analysis there is strong evidence that higher income is associated with lower mortality rates (see fig.1,2).

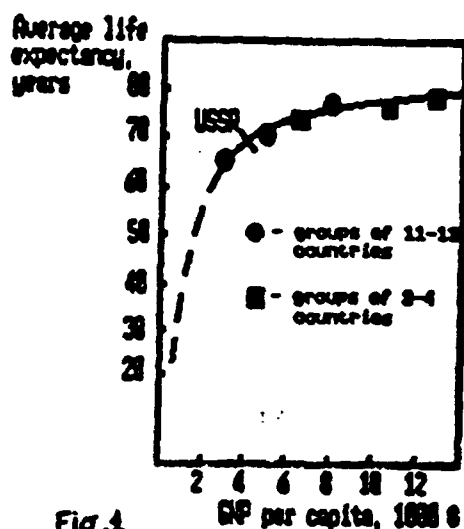


Fig. 1  
Dependence of life expectancy (average in group of countries) on average GNP per capita (in corresponding group). [6]

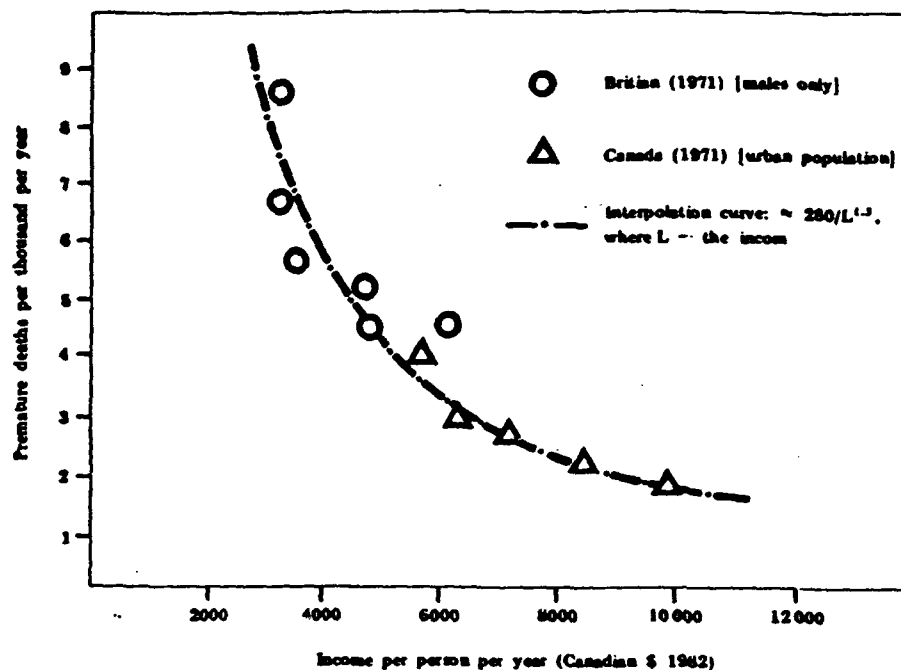


Fig. 2 Mortality and average income in Britain and Canada [7].

These figures may prove the important premise that a society uses its gross national product in manners that reduce health and safety risks for its members and therefore reduce fatalities. Moreover, the above graphs suggest that the risk level of a given society  $R$  which may be defined for instance as the number of premature deaths per thousand per year is exponentially dependent of GNP per capita (which we below denote as  $C$ ). This leads to another important consequence: diminishing efficiency of risk reduction. This fact is illustrated in Table 1. The derivative  $dR/dC$  shows the efficiency of risk reduction and inverted ratio  $dC/dR$  henceforth represents life saving cost. In the same manner from Fig. 1 we may get the amount of money which is required to prolong life expectancy by one year, life prolongation cost. This is an important index of efficiency of socio - economic system.

In fact the dependence  $T(C)$  is not so simple as shown in Fig.1. There are many countries which do not fall into  $T(C)$  curve. Therefore life prolongation cost has special significance as well as gross national product.

country	individual consumption in 1985, \$	GNP per capita in 1985, \$	life saving cost, \$*1000	life prolongation cost, \$*1000
China	1114	2444	30 - 170	0,7 - 4
USSR	2198	4996	135 - 900	3 - 20
Hungary	2971	5765	270 - 1200	6 - 30
GB	5174	8655	950 - 3000	25 - 75
USA	8542	12532	3000 - 7000	75 - 180

*Table 1. Efficiency of socio - economic sphere for different countries.*

Gross national product per capita and individual consumption represented in international dollars were calculated using methodology based on recent UN research. Various difficulties arising when different countries be compared on international dollar base are discussed in [8].

### 3. ENVIRONMENTAL PARITY.

Now we have a very simple model of a society. Specifically we state that all resources produced in the society are apportioned in manners that reduce the aggregate risk level in the society most efficiently. In fact the uncertainties involved in regression analysis of  $R(C)$  for different countries [6,7] suggest that this could hardly be true. Anyway all we need is to place all countries on the same  $R(C)$  curve. Below we designate the aggregate risk level  $R$  as internal risk  $R_i$  to discern it from external risk.

### 3.1 EXTERNAL RISK

Let us suppose that at certain moment a country begins to feel risk from some external source of hazard  $R_e$ . Say, a new power plant has been built on the neighbouring territory or radioactive wastes have been deposited in shallow sea in territorial waters of a neighbouring country. These are typical examples of the transboundary risks.

This risk  $R$  has its own characteristic of risk reduction - monotonously decreasing function of investments in corresponding source of hazard  $R(I)$ . The efficiency of risk reduction may be constant or diminishing (as shown on fig. 3).

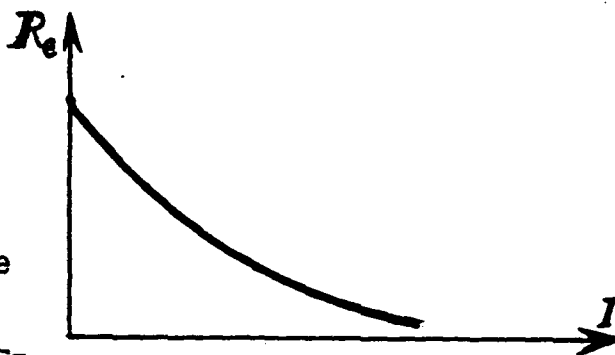


Fig. 3 Efficiency of investment in external risk reduction.

Then the total risk level in the country is the sum of two components:  
 $R_t = R_i(C) + R_e(0)$ . The question arises: should the country invest its (limited) resources in reduction of

this external risk? Consider what if a small fraction  $C$  of its gross national product be invested in external source of hazard. Then the resulting variation of total risk is

$$\delta R_t = R_i(C - \delta C) + R_e(\delta C) - [R_i(C) + R_e(0)] = (dR_i/dC - dR_e/dI|_{I=0}) * \delta C.$$

If  $\delta R < 0$  then it sounds reasonable to invest in the external source of hazard. Otherwise, if  $\delta R > 0$ , it is more profitable for this country (let us say country A) to make the country - owner of this hazard (let us say country B) invest into this source of hazard its own (B) resources! This way the internal risk of country A does not increase whilst the external risk  $R(I)$  does decrease.

At the same time this solution may not be acceptable for

country B if, say,

$$\delta R_t^{(a)} = dR_t / dC^{(a)} - dR_t^{(a)} / dI \Big|_{I=0} > 0.$$

Strictly speaking, denotification  $R$  is not correct because country B is the owner of this hazard. It might have been better to write  $R$  to discern this new risk from the "status quo" risk  $R(C)$ . But for uniformity of designations we prefer to write this risk as  $R$ .

### 3.2 RADIOACTIVITY CLEANUP.

One important consequence may be drawn from the above consideration. Namely, it sets the criterion for risk management. In order to decide whether to invest in the external hazard two derivatives must be compared. The first is reversed life saving cost - the characteristic of the country itself. The second is efficiency of (the external) risk reduction. It characterizes the risk and the technology used for its reduction. For instance, if external hazard is radioactive contamination then  $dR/dI$  characterizes a method proposed for cleaning up radioactivity. Given the life saving cost and the relationship between equivalent dose and individual risk (according to present - day knowledge 1 man\*rem gives  $(2 - 7) \cdot 10$  deaths from cancer and genetic diseases) one may set up the following criterion of investment effectiveness for different countries:

country	The cost (\$) of reduction of equivalent dose on 1 man*rem should not exceed
China	10 - 100
Russia	30 - 600
GB	200 - 2000
USA	600 - 5000

*Table 2. Optimal investments in radioactive clean up.*

We see that a method proposed for cleaning up radioactivity may be acceptable for one country and at the same time absolutely catastrophic when applied in the other country.

Thus we encounter a typical conflict situation between two (or more) countries.

### 3.3 HIERARCHY BETWEEN COUNTRIES.

Theory of games with non - antagonistic interests considers situations of this kind [9].

What do we mean by non - antagonistic interests? Here it implies that for some reasons military, political or economic pressure is unacceptable as means for settling down transboundary pollution conflicts. In fact, this pressure would have increased the total risk for both countries. Not long ago this fact has been widely recognized. Having said this we need to make an important assumption: namely that there are no hierarchical interdependence between countries. It means that all countries are somewhat equal; there is no reason to sacrifice the safety of one country in favour of another.

From mathematical point of view this assumption may be formulated as follows: The goal of risk management for country A is to achieve a conditional minimum of its total risk  $R_t^{(A)}$ . The condition seems quite obvious: it could be *symmetrical* for both countries, because the goal of risk management remains the same for country B.

### 3.4 OPTIMAL INVESTMENTS.

Here we provide mathematical solution to the problem of optimal allocation of resources in risk reduction. We consider the resources (GNP) of both countries  $C^{(A)}$  and  $C^{(B)}$  as constants. Let us suppose that both countries invest part of their (limited) resources  $I^{(A)}$ ,  $I^{(B)}$  into reduction of their "external" risk. Because they invest these parts in the same source of hazard the resulting total risks will be:

$$R_t^{(A)} = R_i(C^{(A)} - I^{(A)}) + R_e(I^{(A)} + I^{(B)}) \quad (1)$$



$$R_t^{(B)} = R_i(C^{(B)} - I^{(B)}) + R_e(I^{(A)} + I^{(B)}). \quad (2)$$

Note that characteristics  $R_e^{(A)}(I)$  and  $R_e^{(B)}(I)$  may be different while the characteristic  $R_i(\arg)$  is the same for all countries.

The best collective agreement  $\tilde{I}^{(A)}, \tilde{I}^{(B)}$  may be derived from the following system of equations:

$$\begin{cases} dR_e^{(A)}/d(\tilde{I}^{(A)} + \tilde{I}^{(B)}) = dR_i/d(C^{(A)} - \tilde{I}^{(A)}) \\ dR_e^{(B)}/d(\tilde{I}^{(A)} + \tilde{I}^{(B)}) = dR_i/d(C^{(B)} - \tilde{I}^{(B)}) \end{cases} \quad \begin{matrix} (3) \\ (4) \end{matrix}$$

Indeed, equation (3) gives minimum to (1) as function of  $I^{(A)}$  and at the same time equation (4) gives minimum to (2) as function of  $I^{(B)}$ . That is why this solution is called non-

improvable. Again we must stress that each country pursues its own "egoistical" purposes. But the environmental parity proves to be the best solution for both countries in the modern highly interdependent world.

If, say, the condition (3) can not be met i.e. after formal resolution of the system  $I^{(A)}$  appears to be negative or non-existent then we must suggest  $I^{(A)} = 0$ . In this case country A should not invest in the external hazard at all. (It may also appear that both countries should not invest in the external hazard.)

Note. This theorem can easily be generalized for N countries which experience risk from the one "external" source of hazard. In this case instead of the system (3),(4) we must solve the similar system of N equations.

#### 4. DISCUSSION.

It is not so easy to grasp all behavioral change that result from the concept of environmental parity. Let us first consider the following case:

#### 4.1 RICH COUNTRY EXPERIENCES EXTERNAL RISK FROM POOR COUNTRY

If  $|dR_i^{(A)}/dC_i^{(A)}| > |dR_e^{(A)}/dI|$  and  $|dR_i^{(B)}/dC_i^{(B)}| < |dR_e^{(B)}/dI|$ ,  $I=0$  then poor country A does not invest in its "external" source of hazard (which is actually situated on its own territory) while country B does invest in this source. Consider a hypothetical situation. Say, Finnish Government voluntarily and irrevocably install gas scrubbers in Karelian power plant. Incredible! Should we consider this as an act of charity? Not at all. Simply it proves to be the most efficient way for Finns to allocate their own resources in reduction of their own risk level.

#### 4.2 EFFICIENCY REDUCTION AND OPENESS.

The following situation might indicate the need for environmental parity adoption. Suppose a highly developed country manages its risks most efficiently. Then upon the time it gradually "slides down" the curve on Fig. 2. Due to efficiency reduction the safer country becomes the slower risk level goes down. At a certain moment it becomes clear that instead of asymptotically approaching zero (as shown on Fig. 2) the risk level approaches the certain limit. The country continues to spend large sums of money in risk reduction but the risk level does not go below this limit! People are dying. It may suggest that an external (and yet unmanageable) risk appears on the scene (see Fig.4).

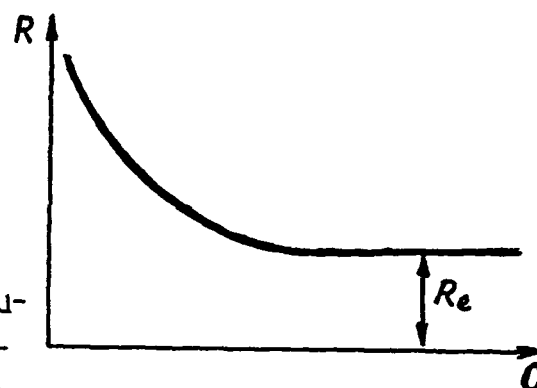


Fig.4 Efficiency reduction and external risk

In order to manage this external risk a special international institution must be established. And the information on all existing in different countries risks and hazards must be accessible for this institution. This requires a sufficient degree of openness between countries. For the more

precise and exhaustive information a country submits the more profitable and effective for this country the "non-improveable collective agreement" is (or may be reached). Any misunderstanding or secrecy only diminish the effectiveness of cooperative agreement of this kind.

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## SOME ASPECTS OF RADIOACTIVE WASTES DUMPING INTERNATIONAL REGULATION

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### INTRODUCTION

Development of atomic and nuclear energy has created a number of new problems the mankind faces now. The problem of radioactive wastes (RW) isolation from biosphere is among them. This problem is of such importance that today it determines the development of nuclear energy (21).

Nowadays the humanity realises that wrong or irresponsible decision in this field can lead to unreversal disruptions for all the planet.

RW management is not only technological problem, it has also socio-economic, psychological, ethic aspects. The legal regulation of RW burial has a paramount importance. The emplacement of enormous volumes of low- and intermediate-level wastes is developed better at national level, rather than at international one. Different countries have accumulated considerable experience in this field, which is different for each country because of its specific economic, natural and political conditions. This report is not aimed at elucidation of this enormous field, we are going to pay attention here to international aspects of RW dumping into the ocean.

Currently people are more and more coming to a conclusion that the ocean is in common property and it could be protected from radioactive contamination only through states cooperation and consultations. All countries are interested in developing of common regulations on RW treatment and disposal and there international legal regulation.

## INTERNATIONAL LEGAL REGULATION OF MARINE RADIOACTIVE WASTES DISPOSAL

The Geneva Convention on the High Seas adopted at the first U.N. conference on the Law of the Sea (UNCLOS I, 1958), was the first multilateral agreement dealing with RW dumping at sea. According to it every state shall take measures to prevent pollution of the seas from the dumping of RW and other harmful agents. All states shall cooperate with the competent international organizations, taking into account any standards and regulations which may be formulated by them (1).

In this way, as it noted in (2), the Geneva Convention contains only general prohibition of radioactive contamination of the seas and atmosphere. Its standards were too unconcrete. In (3) it is mentioned that absence of burial's distinct prohibition in the Convention has led to the fact that it couldn't have had been a source for states' clear obligations.

In an accompanying resolution UNCLOS I also recognized the need for international regulation of the dumping of RW in the ocean, noted the recommendations made by the ICRP on human dose and environmental concentrations of radioactivity and recommended that the IAEA pursue whatever studies and take whatever action is necessary to assist States in controlling the discharge or release of radioactive materials to the sea, in promulgating standards, and drawing up internationally acceptable regulations to prevent pollution of the sea (4).

The major international agreement on RW dumping is the London Dumping Convention signed in 1972 and come into force in 1975 ( for USSR in january 1976). It is devoted to intentional burial of harmful wastes throughout all the ocean except the internal waters.

According to Art.1 the London Dumping Convention (LDC) parties shall undertake all practical measures to prevent potentially dangerous for human health or marine life dumping (5). In accordance with the LDC high-level wastes dumping is absolutely prohibited ( Annex I ). Special Permits must be issued by national authorities for the dumping of other radioactive

wastes, which is listed in Annex II. In issuing permits national authorities are to give careful consideration to factors specified in Annex III, including prior studies of the dumpsite. For RW included in Annex II, parties should take all recommendations of the competent international organization - IAEA. Parties should notify the International Maritime Organization (IMO), which has been designated the secretariat of the Convention, of the permits they have issued.

The IAEA is designated to determine high-level wastes unsuitable for dumping, to adopt dumping standards, to give recommendations to national authorities. The LDC guarantees that parties of the agreement will in full account take into consideration the IAEA's recommendations.

The LDC can be called historical without any exaggeration. Nevertheless it has a number of substantial shortcomings. D. Hodges (8) emphasized that its large defect is absence of prohibition of radioactive and other harmful wastes disposal at sea by those countries who didn't join the Convention.

The LDC is not designated on "accidental" dumping, on marine effluents, releases from nuclear facilities and naval vessels.

It is mentioned in (7), that the LDC doesn't give the distinct definition of low-level wastes and the IAEA's definition is also not concrete. This uncertainty is shared by scientific public opinion which has no general point of view with respect to low-level radionuclides consequences.

One more shortcoming of the LDC is that the state which is going to dispose of RW has to give the permit to himself, i.e. coercion and punishment are entirely in national authority's competence.

Authors of (11) stress that absence of responsibility for its requirements violation is thought to be the main Convention's defect. Art. X refers to international law principles concerning with state responsibility for damaging the other states environment or any other environment zone resulting from wastes or any other materials discharge. The agreement to develop the procedure for responsibility determining and debates regulating is expressed.

It has become obvious now, that the question of responsibility for radioactive contamination is quite undeveloped

in international law. In this particular case the matter is complicated by many factors: the dangerous consequences of radiation could be discovered in large period of time (more than 20 years), there link with nuclear accident could be hardly determined.

The Consultative Meetings of parties to the Convention are designated to be held at least in every two years. The 7th Meeting 1983 adopted a resolution that appealed to stop radioactive dumping. In 1985 another resolution on voluntary refusal from RW dumping was adopted. It requires the careful assessment of radiation impact on marine environment and people.

RW dumping at sea is the subject to several regional convention - Helsinki, 1992; Paris, 1992; Barselone, 1976; Bucharest, 1992. They control dumping in the certain areas: The Mediterranean, North Atlantic Ocean and North Sea, Black Sea, Baltic Sea.

The U.N. Conference on the Environment and Development, 1992 offer to come from voluntary refusal of RW dumping to the prohibition of such practice (23). The conference's approach is based on precaution principle.

So, the international agreements' analysis brings us to the conclusion that conventional standards pronibiting radioactive pollution of marine environment do exist in international law. But, as it is noticed by Malinin (18), there efficiency depends on there concrete character and needs further development.

#### FOREIGN RW MARINE DISPOSAL PRACTICE

RW dumping into the ocean practice has already more than 30-years history. The first burial was made by USA and took place in 1946 in notheastern part of the Pacific Ocean. Up to 1970 USA have disposed of about 107000 containers with total radioactivity  $4300 \cdot 10^{12}$  Bq in the Pacific and Atlantic oceans, mostly at four sites. In 1970 dumping from USA territory was stopped, which can be explained by economic and socio-political reasons (24), and is not resumed yet. Nevertheless, this policy is being reconsidered

now because of population's protests and accumulation of large volumes of wastes.

Europeane countries begun RW dumping at sea in 1949. Up to 1967 the UK was the only country, disposing of its wastes in sea. Then the necessity of general Europeane coordination on low-level wastes dumping has risen. Nuclear Energy Agency (NEA) took the role of such a coordinator. Till year 1967 RW dumped in the ocean were not registred internationally. There total volume roughly estimated equales 100 000 tons with activity  $30000 \cdot 10^{12}$  Bq.

In some years Belgium, Italy, the Netherlands, France, Germany, Switzerland and Sweden joined the UK. Up to 1974 RW were released to different areas of the Atlantic ocean. Since 1974 the only site in notheasten Atlantic had been used for this purpose.

Japan was emplacing RW into sea since 1955, another country who was disposing of radioactive substances in that region since 1968 up to 1973 is South Korea.

Public opinion had a noticeable effect on ocean's use for RW releases. In the latest 1960s the movement against environment contamination, the marine in particular has acquired the organized international character. This result in some global agreements adoption. The London Dumping Convention is among them. This convention has led to progres in dumping at sea explorations.

The latest officially registred burial in the ocean ( except USSR and Russia) took place in 1982 (23).

In general since 1946 to 1982 RW with total activity about  $46 \cdot 10^{15}$  Bq ( $1,24 \cdot 10^6$  Cu) were dumped into the ocean as it was pointed out by the IAEA. This figure does not include USSR and Russian dumped wastes which weren't notified to any international organization and liquid wastes released into sea from nuclear fuel reprocessing plants (23).

#### DUMPING PRACTICE IN USSR AND RUSSIA

The dumping practice in USSR and Russia was an absolutly secret subject up to now. By the order of russian Presedent the



Governmental Commission on the issues dealing with radioactive dumping at sea was created in 1992. Here we'll use its report in very brief description of this problem. This report provides only very approximate assessment of some experts meanwhile the detailed information on dumped wastes composition and volume as well as radiological situation in the vicinity of dumpsites is still unknown.

The very first RW releases into sea took place in White Sea in 1959 in accordance with "Lenin" atomic ice-breaker tests. In 1960 the practice of regular releases of liquid wastes begun, in 1964 the burial of solid materials with high, intermediate and low activity came into practice in nothen and far-eastern seas. It had irregular and uncontrolled character.

The matter of particular concern is nuclear range at Novaya Semlya, where enormous ammount of containers with RW, submarines were dumped without any control. Now it is impossible to estimate nuclear explosions impact on environmental radioactive contamination and to give the truth description of ecological situation in 150-km archipelago's zone (20).

Since the LDC coming into force USSR has undertaken a number of measures aimed to international rules and obligations fulfilment. In 1979 the Council of Ministers passed the decree "On measures aimed to provide fulfilment of Soviet obligations resulting from the LDC".

The releases from Murmansk Lines were reduced and then totally stopped. But the Navy didn't take measures to stop discharges because of lack of technical facilities, depositories and misunderstanding the doctrine of "sovereign immunity", which enables not to consider dumping from naval vessels as the LDC violation.

The Commission brought to notice the fact that RW dumping from the USSR territory was twice illegal:

- national standards and requirements didn't meet the requirements of the international agreements recognized in the USSR;
- even these documents were violated.

It should be stressed particularly that the high-level RW dumping prohibited by the LDC had been conducted under single permission of the Naval and Ministries.

The total activity of all dumped RW in nothen and far-easten seas could be estimated as  $325 \cdot 10(3) \text{ Cu}$  ( $12 \cdot 10(15) \text{ Bq}$ ). The experts gave the upper limit of dumped wastes activity as  $2,5 \cdot 10(6) \text{ Cu}$ .

The USSR dumping practice was prolonged by russian Navy. In spite of rather insignificant dumped RW activity ( $55,2 \text{ Cu}$  or  $2050 \cdot 10(9) \text{ Bq}$ ) Russia faces the fact of the LDC violation.

The fulfilment of russian international obligation will require the following measures:

- 1) notifying the secrstariat of IMD and the IAEA of dumped RW;
- 2) dumpsites exploration under support and participation of competent international organization and all states, which are interested in the matter;
- 3) organization of effective monitoring of high-level wastes dumping;
- 4) development of sea cleaning from high-level wastes, which are ecologically dangerous;
- 5) immediate providing of treatment and safe storage of RW from nuclear vessels.

#### DISAGREEMENTS ON RW DUMPING INTO THE OCEAN

Large disagreements in RW dumping problem evaluation exist both within the countries and at the international level. They make the united RW management policy development harder.

Several groups of such disagreement could be outlined:

- positions of different countries, some of them intend to dump RW into the ocean, meanwhile the others demand there absolute prohibition;
- conflicts between the population concerning on possible consequences of the burial and national authorities who plans such burial;
- contradiction between the population and national or transnational companies (for example, people's fear to consume fish or seafood trawled near the dumpsites, which presents great

losses for the companies) or other socio-psychological aspects of possible consequences of RW transboundary transfer;

- different interpretation of legal standards by civil and military structures;

- different positions of international organizations based on various concepts ( the IAEA - Greenpeace);

- essentially different and some times contrary scientific points of view on RW dumping at sea.

The last group of contradiction is thought to be fundamental, resulting to some certain extent in all the other disagreements. Therefore let us consider it more detailed.

At present scientists haven't come to the single opinion on radiation effect on marine ecosystems and human health. The Group of independent experts from the IAEA and International Council of scientific societies elected at the request of the 7-th Consultive Meeting of the IJC for additional scientific assessment of dumping impact hadn't discovered any reasons for canceling sea dumping if only all internationally adopted principles of radiological protection are observed (10). They also concluded that individual doses for marine organisms as well as for people are small enough that enables to consider sea dumping to be an appropriate practice.

On the other hand many scientists insist on intolerance in question of RW dumping. A group of american researchers (7) gives a number of well proved conclusions resulting from dumpsites exploration:

- 1) dumpsites examining shows the releases of radionuclides from containers into the marine environment;

- 2) the present and anticipated dumping operation impact on the environment and human health was estimated on the basis of incorrect models;

- 3) dumpsites examining shows that radioactivity from dumpsites enter the ocean food chain and is being discovered in edible fish;

- 4) the existing international radiological standards haven't been revised since the 1950s, now they are being revised taking into account that cancel threat and other harmful radiation effects are much serious than it was thought;

- 5) land burial of RW is more preferable than sea dumping for

a number of reasons: it is more profitable economically, containers keep germetic for a longer period so there isolation from the biosphere is guaranteed for a longer term.

Greenpeace experts also believe that there are enough reasons for sea dumping prohibition. In there opinion the main criterions for the way of RW burial should be:

- the possibility to watch the burial consequences;
- the ability to prevent possible negative processes.

6) dumping legal state is rather doubtful.

At present it is dictinctly defined that one-sided dumping is unacceptable and prior international consultation is necessary as it is required under the LDC. Nevertheless the detailed legal regulation of sea dumping is not established yet.

#### WHAT SHOULD BE DONE IN SEA DUMPING INTERNATIONAL REGULATION

Nowadays it is widely recognized that no one state is protected from pollution while the problem of areas of common use wouldn't be solved.

As it was mentioned above recently the sea dumping matter are becoming more and more the political problem. Political conflicts and above- mentioned disagreements make the conciliation achivement more difficult. In our opinion the following measures adopted at international level are of prime importance:

- Effective monitoring in dumpsite areas is badly needed.
- Complete register of dumped RW is necessary for dumping impact assessment and for the LDC guaranty. All the significant amounts of RW released into sea or rivers running into sea should be registred carefully. This information should enter the international data center which should be available for all countries.

- In spite of a number of conventions the most part of the oceans is still open for radioactive pollution. Present international law standards are too unconcrete and effectiveless.

The responsibility for radioactive damage issues are out of legal regulation.

- The general international convention on all kinds of RW dumping prohibition is badly needed. It should take into account the measures against all possible sources of radioactive contamination.

- The international program on radioactive pollution prevention is of great importance. It should consider not only scientific and technological aspects but also political, legal and socio-economic problems. The effective international control of all sources of ocean radioactive contamination should be developed and entered into force in the frame of this program. The question of particular concern is the involvement of military activities in agreements process. This secret field of activity should be under the strict control in this program.

Conference at the Woods Hole Oceanographic Institution on  
'Radioactivity and Environmental Security in the Oceans  
Research and Policy Priorities in the Arctic and North Atlantic'  
(June 6-10, 1993)

ENVIRONMENTAL SECURITY AND GOVERNANCE ISSUES IN THE ARCTIC OCEAN  
by Dr Renat Perelet, Research Leader,  
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The purpose of this paper is to present suggestions on possible directions of research on issues of governance and environmental security in the Arctic that should contribute to dealing with radioactivity and other major environmental problems in the Arctic. It starts with stressing radioactivity threats as most urgent ones affecting national and international security and leading to international tensions and likely conflicts. Then it briefly discusses what is available, adequate and missing in current international socio-economic and legal research programmes. It is suggested that a research programme in the long run should lead to the design of an international environmental security regime in the Arctic ocean and a treaty of the Arctic to achieve sustainability in this region.

#### Environmental Threats in the Arctic

Evidence has been growing of radioactive and other environmental threats to the countries in Northern hemisphere and the Arctic area resulting from past and present environmentally unsound human activities (1). Some 30 nuclear dumpings or accidents were cited that mainly related to military industrial and naval activities. Meanwhile, accidents involving US and Russian nuclear powered navy vessels are still continuing (e.g. the one on 25 March 1993 in the Barents Sea).

It was proved that the former USSR had exercised radioactive waste dumpings with various levels of radioactivity starting from 1959. The Soviet navy and the ministry of shipbuilding were mainly responsible for that activity. The former continued liquid radioactive waste disposal into the Barents sea in 1991-1992. Neither the International Marine Organization (IMO) nor the International Atomic Energy Agency (IAEA) was notified in this connection. The largest threat is believed to come from dumpings of nuclear submarine reactors and an assembly with nuclear fuel from the nuclear powered ice-breaker 'Lenin' into shallow waters of the Kara sea off the Novaya Zemlia archipelago. The adoption of the policy on nuclear waste disposal into the ocean led to abandoning plans to build solid and liquid nuclear waste treatment plants in the USSR as well as to the lack of monitoring of radioactivity in the Arctic seas during more than 25 years which makes it now impossible to determine limits to solid radioactive waste disposal, rates and scale of radionuclide releases (2).

The consequences of viewing the Arctic ocean as a waste sink, e.g. for radioactive waste disposal and nuclear weapon tests, can foreclose its use as a source of environmental assests, including useful minerals, energy, transportation, telecommunication media, fresh water supplies, tourism, notwithstanding the severity of Arctic operating conditions.

At present the exploitation of arctic resources along with the air and marine pollution especially by the military are major threats to ecological equilibrium in this region. More than two thirds of Russian gas deposits, one third of US oil deposits and unassessed deposits in Canada are in the Northern Arctic (3).

Biological processes under low temperatures in the Arctic take place slowly with the biomass growing at one per cent on average of the production in temperate areas. The polar regions contain some of the most fragile ecosystems. Indiscriminate exploitation of their living resources has already brought some mammal species to the brink of extinction which may have caused a serious imbalance in the natural ecosystems (4).

In addition, the ocean becomes a source of international tensions which may lead to conflicts. Arctic shipping has been an issue between the United States and Canada for some time (5). This issue may be exacerbated if it becomes a multilateral one.

#### Security Dimension of Arctic Environmental Problems

Many of these problems threatening humans and vulnerable Arctic marine ecosystems affect the core of national interest, namely national security, and can be considered as its essential part, i.e. environmental security in which the concern over the Arctic radioactive pollution (RAP) may become a cornerstone. The appropriateness of using this concept is increasingly supported by researchers (6).

In fact, three major Russian national interests in the Arctic are linked with security considerations. They are the exploitation of off-shelf oil and gas deposits discovered in the 1980s, the use of the northern sea faring route linking Europe and south-eastern Asia, and, last but not least, the deployment of strategic ship-borne nuclear missiles in Russian coastal waters of Arctic seas (7).

Environmental security considerations should also be taken into account in this strategy.

In the above context, environmental security may be viewed as the protection of sustainable performance of humans and societal institutions from risks of (a) dwindling environmental assets and/or access to them, (b) adverse effects of environmental changes, e.g. RAP, on human health and well-being, and (c) environment related intra- and international tensions and conflicts. Thus, the first two components deal with relationships between humans and nature, while the third - among humans, in particular among states, over the environment.

The growing awareness of ecological interdependence, environmentally sound technological (EST) change, and

increasing public pressure can be viewed as driving forces for developing concerted international efforts to combat radioactivity pollution in the Arctic. This approach was tested in developing international cooperation on air pollution (8).

The first element requires better understanding with the help of natural scientists of ecological systems and processes in the Arctic due to natural and human-made activity (in particular related to RAP) as well as the sharing of responsibilities and the definition of liabilities of actors in the Arctic (a role for social sciences). The second should lead to strict application of the precautionary principle in the development and use of technology in the area and incentives for the international transfer of EST. That implies relevant changes in environmental regulations and economic mechanisms, production and consumption patterns to stimulate EST change. The third element deals with environmental values, education and training, environmental risk perception and communication, popular participation in decision-making and implementation, political mobilization, institutional change to combat RAP.

The issue of RAP in the Arctic and its human dimensions should also be considered in the context of global environmental change. The radioactive waste disposal (RAWD) and other nuclear related activities in the Arctic were largely considered safe in belief that slow environmental processes would impede wide and rapid dispersion of waste. The perspective of global climate warming and attending sea level rise may stimulate access to Arctic mineral resources, marine transportation, increased human settlements, tourism, and economic activity along the northern seashore. These issues would present an environmental security challenge for cooperation or/and, if unresolved, may lead to international tensions and conflicts.

#### Arctic Environment in International Programmes

What international research programmes are available to address Arctic radioactivity and RAP problems? There are few international programmes that cover the Arctic environment, for example, an eight-nation cooperation programme in the Arctic with International Arctic Scientific Committee (IASC) or some WMO and IOC activities. None of them specifically addresses issues of radioactive pollution. IAEA is to launch a project to assess RAP in the Arctic seas. In fact, about 80% of research in the polar areas has up to now been devoted to the Antarctic and only 20% to the Arctic region. This proportion should be reconsidered.

It is notable that at the United Nations Environment Programme (UNEP) the Arctic as an emerging environmental problem area first appeared at the 15th session of UNEP's Governing Council in 1989. A brief report with an oblique reference to radio-active pollution (RAP) was presented and merely taken note of at the 16th session of UNEP GC in 1991 (9). UNEP's regional seas programme does not include the Arctic. UNEP's document on Strategies for the protection and development of the oceans and coastal areas (10) prepared in the wake of UNCED does not mention either the



Arctic or the RAP subject.

Marine land-based pollution by radioactive wastes is indicated for the North Sea and in North America regions (11). Radioactive pollution is briefly mentioned in the State of the Environment Report to UNEP's GC.16 (12). A core project - START - of the International Geosphere-Biosphere Programme (IGBP) singled out the Arctic region as a key area for research on global change.

The UNCED's Agenda 21 in its chapter 17 'Protection of Oceans' does not go to the level of vulnerable Arctic marine ecosystems nor does it discuss specific radio-active marine pollution there. One finds no references in this regard in 'the World Environment 1972-1992' prepared by UNEP (13) or in 'A Nordic Environmental Research Programme for 1993-1997' (14).

The following major international environmental agreements on marine pollution are available to address Arctic RAP problems:

1972 Convention on the Prevention of Marine Pollution by Dumping of Wastes and other Matter (London Dumping Convention) - IMO;

1973 - International Convention for the Prevention of Pollution from Ships and the Protocol of 1978 Relating Thereto with Annexes (MARPOL 73/78) - IMO;

1982 - UN Convention on the Law of the Sea (LOS Convention) - UN Office of Ocean Affairs and Law of the Sea - OALOS;

1990 - International Convention on Oil Pollution Preparedness, Response and Co-operation - IMO (13).

The above indicated legal and institutional means are clearly not adequate enough to achieve sustainability of the Arctic region. All this may make one to believe the RAP in the Arctic and North Atlantic is a missing issue on the international environmental agenda.

#### Governance Issues in the Arctic Ocean

Environmental changes over the last decades due to radio-active marine pollution in the Arctic ocean has only recently been recognized as presenting an emerging environmental problem of global significance that needs to be properly managed.

Management in the Arctic implies use: rational use. Emphasis on the concept of integrated management reflects a critical stage from the earlier views of most environmentalists that the oceans had to be preserved unchanged rather than used rationally (13). If one can ban economic activity in the Antarctic it is hardly possible and reasonable in the Arctic.

The policies, measures and arrangements for the protection and use of the marine and coastal environment, if they are to be rational and thus effective in the long term, should be based on an improved understanding of marine and coastal ecology, including ecosystem dynamics; of the relevant ocean processes; and of their interaction with terrestrial and atmospheric systems (14).

However, proposed strategies as seen from documents prepared for UNEP's GC 17th session do not specifically deal with the management of radio-active waste that have already been accumulated in the oceans and, in particular, in the Arctic area.

Some kind of governance in the area seem to be needed and appropriate research should be carried out on outstanding issues.

Is the Arctic an international commons? Is the Arctic an international commons shared by or divided among northern circumpolar nations? Should its problems be handled by northern circumpolar nations only or with a broader international community be involved? Should an international regime be worked out for the Arctic? International multilateral cooperation needs in the Arctic are now well recognized. Should UNEP expand its regional seas programme including the Arctic area? What should be the role of non-governmental organizations, including grass-root, scientific and business communities? Non-governmental organizations were prominent in stimulating public concern over and governmental activity on radioactive pollution by the military in the Arctic seas.

#### A Treaty of the Arctic?

Research into environmentally sound management of the Arctic, economics of the Arctic (including valuing Arctic environmental assets and conducting environmental accounting of its 'natural capital', designing appropriate economic mechanisms for the Arctic), environmental security issues in the Arctic can be crowned by or oriented towards the design of a treaty of the Arctic in which radioactivity issues should be highlighted. An international regime for the Arctic and a treaty of the Arctic should be observed also by the military which may require appropriate institutional structures. The UNCED followup process should include the Arctic as a priority item.

The major RAP issues to be handled in the treaty of the Arctic should relate to pollution from land-based manufacturing facilities dealing with radioactive substances, nuclear weapons testing ranges, naval and civil vessels using nuclear fuel or transporting radioactive substances, nuclear waste dumped into Arctic seas.

Provisions should be made for national and international radioactive risk management to achieve environmental security and cover routine operations and accidents and include risk analysis, assessment, economics, communication, perception as well as provisions for appropriate training and education.

Control over the implementation of such a treaty should include strict liability of its violators and inspection facilities.

Non-governmental organizations should play a prominent role in governmental decision-making on environmental security and governance issues. Action to ban, limit, phase out and/or raise safety of radioactive operations should duly be justified and openly made.

The expertise of the following international organizations and research programmes can be useful to complement national efforts: UNEP, WMO, IAEA, UNESCO, NATO, IUCN, IGBP, the Human Dimensions of Global Environmental Change programme (HDGECF) and others.

Within the above context, an international research programme on Environmental Security in the Arctic can be designed to work out a set of measures leading to an Arctic environmental security regime that would include natural, engineering and social scientists. As a first phase, a set of measures should be

elaborated to combat radioactive pollution in the Arctic.

The three levels of activity can be considered:

(a) a short-term programme to deal with nuclear weapon testing environmental impacts and radioactive waste disposal (RAWD) sites (their identification, RAWD site rating according radio-activity release, national/international action to control RAWD sites) and to monitor Arctic radio-activity (internationally agreed measurement parameters, methodologies, instruments, intercalibration procedures); (b) a middle-term programme to manage RAP (marine and air pollution) through, in particular, raising integrated safety of technologies used in the Arctic ocean such as nuclear powered vessels, nuclear test facilities, etc. and controlling RAWD, and (c) a long-term programme to review the use of radioactive substances, nuclear based technologies and RAWD in the Arctic ocean.

In conclusion, as an introduction to the above programme a project proposal on 'The Development of a Plan of Action to Combat Radioactive Pollution in the Arctic Ocean' is briefly described.

Long-Term Objectives: to achieve rational use of unique Arctic ecosystems, to improve health of human habitats in the Arctic. Short-Term Objectives: to determine principles of and approaches to preventing the Arctic environment from radioactive pollution and to elaborate a plan of action therefor. This project is now being discussed in Russia. If an international project is conceived other participating parties can only enrich it with their views and inputs.

Background: The Russian Arctic area, especially its western part, abounds with concentrated sources of elevated nuclear and radioactive threats (the Novaya Zemlya testing range, nuclear submarine bases, nuclear powered ships, radioactive waste disposal sites, seashore based nuclear material handling enterprises, ect.).

The Russian government policy to achieve nuclear and radioactive safety is at its early stage. Systems studies of the problem in the Arctic has not been carried out while they are urgently needed to be based on the information from the Russian foreign ministry, the State Committee for the North, the Ministry of Defence (the Navy).

Expected Outputs: a draft plan of action to combat radioactive pollution in the Arctic Ocean; recommendations as regards legislative acts to be adopted to prevent the Arctic from radioactive pollution; recommendations as regards further development of international cooperation to protect the Arctic from radioactive pollution.

Duration of Project: one year.

Cost of Project: 5 m/y + two month field studies.

Project Activities:

1. Analysis of major sources of nuclear and radioactive pollution in the Arctic.
2. Assessment of the Arctic ocean radioactive waste pollution.
3. Assessment of contribution by shore based naval bases to radioactive waste pollution.
4. Assessment of nuclear powered transportation as a source

of radioactive danger, of its international and national legal control.

5. Impact of the Novaya Zemlya nuclear testing range on radioactivity levels and international relations in the Arctic.

6. Analysis of interests of different Arctic related states as regards nuclear powered transportation development and radioactive waste management in the Arctic.

7. Elaboration of a draft plan of action.

Suggested Implementing Parties in Russia: Institute for Systems Analysis of the Russian Academy of Sciences (ISA RAC), Institute of World Economy and International Relations, RAC; Institute of Geography, RAC; Scientific Centre of the RF Committee for the North, experts from the Russian Foreign Ministry, the Ministry of Defence (the Navy).

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**NEW DIRECTIONS IN THE CULTURAL AND POLITICAL DEVELOPMENT  
AMONG ARCTIC PEOPLES : IMPLICATION FOR RADIOACTIVE  
WASTE DISPOSAL IN THE ARCTIC**

**Key-note address:**

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Much has been written and said about radioactive waste in the Arctic, but there has been remarkably little discussion about the impact on the peoples of the Arctic nor has there been much consultation with them on the issues. Important cultural and political changes are taking place in the Arctic; changes that will have important policy-making consequences, even scientific consequences, and which will necessitate indigenous peoples and other peoples of the Arctic becoming directly involved in the decision-making process in future years.

Not only has little attention been paid to the peoples of the Arctic, there are indeed scientists and policy-makers who continue to discuss the Arctic as a place where no one lives, as literally a wasteland, where waste can be dumped without serious consequences.

Nobody lives there - you still hear it said - well, somebody does live there - approximately 8-10 million people (depending on how you define Arctic). Peoples with their own cultures, languages, histories, and with a new-found political and economic clout. As well, the Arctic is becoming an increasingly complex region with many different stakeholders, each with its particular vested interests and specific needs.

In the 60's and during most of the Cold War period, the only, or at least the main players, were the national governments and the military - nuclear tests could be carried out, and waste could be dumped. National governments closed their eyes, the military was secretive; the information was classified or never recorded because neither West nor East kept particularly accurate records of chemical or radioactive dumping or burial. The time was different, nobody paid too much attention; neither the public nor the media were as well informed as they are now - the Iron Curtain and the "Ice Curtain" were firmly in place.

In the early 70's, however the circum-arctic world started to change - environmental NGO'S such as Greenpeace came into being; mining, oil and gas industries started large-scale projects in the North, and the peoples of the Arctic (particularly the indigenous groups) started a quiet bloodless revolution, for they wanted a voice, a real and serious voice in what, they after all, considered their homeland as well as the source of their livelihood and culture.

The Arctic was no longer (of course, it never was) a place where no one lives. And in the 70's, 80's and 90's, it became increasingly and abundantly clearer each year that people live here who cares passionately about the land, the air and the oceans.

The Arctic has transformed itself over the last 30 years, this transformation has significant policy implications. What we find now in most parts of the Arctic is a lively, modern and very astute society with increasing political and economic power.

Let us consider the Canadian case for a moment. In Canada, literally at the very moment of this conference, the political make-up of our country is changing because our North is changing. What implications will these new realities have for policy directions, for political priorities when neither policy nor priorities will be established by the usual players?

We are in the midst of extraordinary change. Two new bills, bill C-132 and C-133 have just been passed by the Canadian parliament and have received royal assent; these two bills ratify the largest land claim in Canadian history and create a new territory to be called Nunavut, "our land" in Inuktitut.

This new territory, created by dividing the present North West Territories into two, is not officially an ethnic-based territory, but de facto it will be an Inuit territory, as close to 90% of the inhabitants are Inuit. At the ceremony in Iqaluit, when the Nunavut agreements were officially signed on May 25, by then Canadian Prime Minister Brian Mulroney, the Canadian Minister of Indian Affairs and Northern Development, Thomas Siddon, declared that "... settling and implementing the land claim and creating the Nunavut territory, with public agreement, signifies a bold new partnership between Canada and the Inuit of the Northwest Territories, both the creation of the territory and the settlement of the land claim provide the Northwest Territories Inuit with the means to take their rightful place in Canadian society while playing a vital role in the economic and social development of the region."

The newly created Nunavut territory will also have a full-fledged territorial government to be located in a capital yet to be selected. If all goes according to plan, the new government and territory will come into effect in 1999. Along with this new political power, the Inuit will obtain 1.4 billion Canadian dollars in compensation over the next 14 years and receive ownership of 352,191 square kilometers or 135,359 square miles of land within Nunavut; furthermore they will have preferred access to the rest of the land, special rights in regards to waterways, will receive some royalty payments, and will be involved in the management and regulation of all renewable and non-renewable resources. Thus the 17,500 Eastern Arctic Inuit will have a land base, a new territory (approx. a little more than 1/2 the size of India), a new legislature, regulatory powers and compensation money, in other words, major political and economic power.

And much the same pattern is seen everywhere else in the circum-arctic world. The 6,000 Inuit in Northern Quebec (Nunavik) received a large measure of autonomy with the James Bay Agreement in 1975; in Western Canada, in the Western Arctic and the Mackenzie Delta, the (approx. 6,000) Inuvialuit have also received a comprehensive land claims and compensation agreement. In 1972, the approx. 30,000 Alaskan Inupiaq and Yupiq signed the Alaska Native Claims Settlement Act and received a limited form of home-rule government as well as land and compensation money. In 1979, the 42,000 Greenlandic Inuit received broad powers through the implementation of Home Rule government in Greenland. Though Greenland is still very much part of the Danish Realm, it in many ways operates almost like a nation-state; it has full jurisdiction over all matters except defense and foreign affairs. In Nordkallotten, the Sami have established a Sami parliament and a Sami council; the indigenous peoples of the Russian Arctic, the so-called "26 small peoples of the North", are starting to be involved in international collaboration and in environmental causes, though none of them yet has the political and economic clout of the Canadian Inuit, the Greenlanders or the Alaskan Inuit. However, Canadian, Greenlandic, and Alaskan Inuit are now working with the Russian indigenous peoples, particularly in Chukotka.



All of the peoples of the Arctic share common concerns - the major one being the state of the environment. And they have also proven themselves to be astute politicians making their mark on the international scene through organizations such as the Inuit Circumpolar Conference, and in international fora such as the United Nations Human Rights Commission, the International Union of Circumpolar Health (a member of WHO), and UNEP. The peoples of the Arctic are involved in the Arctic Environmental Protection Strategy (the Rovaniemi Process) and in the proposed Arctic Council, to name just a few.

Another major change has been the proliferation of international collaborative efforts including the Arctic. Major initiatives like the Arctic Environmental Protection Strategy, the founding of the International Arctic Science Committee (IASC), the establishment of AMAP (Arctic Monitoring Assessment Program), a large number of co-operative science projects and conferences have become possible in recent years aided by political change, the declassification of information, less restrictive travel and co-operation regulations as well as the general trend to internationalization. Many of these are now directly involve the peoples of the Arctic.

Add to these events, the fact that the number of interested parties in the Arctic has increased so that now the concerns of the indigenous peoples, the military, industry, NGO's, environmental groups, the scientists, non-indigenous residents, regional, territorial, provincial, home rule, national and international governments and agencies all have to be taken into account in varying degrees. The Arctic has become a complex and modern place where the usual rules of the southern establishments no longer take precedence.

How then is the policy-maker, or politician to make sense out of this? It, of course, depends on the policy-maker and on the politician and we have discussed several options at this conference. Several of our Russian colleagues have warned us against the do-nothing policy-maker who will close his or her eyes, will find a scientist - or quote one out of context - who will say that we do not have enough data, that there is absolutely no imminent danger - and if confronted by media accounts, will say "oh, well, you know that the media, even the New York Times, blows everything out of proportion". But as our Russian colleagues have reminded us, this is a dangerous option.

Then there is the more courageous policy-maker or politician - he or she will weigh all of the interest-groups' position including the concerns of the peoples of the Arctic and will ask questions, that we too may ask ourselves as we at the conference here work towards recommendations or principles for a policy:

- should we not always err on the side of caution when the potential dangers are so great?
- should we not operate on the assumption that a worst case scenario might happen - rather than hope for the best case scenario?
- what will all of this cost? and what other programs are we willing to cut at a time of financial restraint?

- how do we deal with the new political realities of the Arctic, both nationally and internationally?
- how do we avoid causing unnecessary fear or panic? how do we communicate realistic concerns to the peoples of the Arctic? how can the mass media be used in this regard? how do we deal with psychological fear? with perceptions?
- how can we improve communications between international and national programs and institutions - so that agencies and programs can work together rather than at cross-purposes?
- what are the possibilities of large-scale health problems, contaminants in the food chain?
- should radioactive waste studies be incorporated into national and international global change program?
- what is the radioactive waste doing to the environment, to us, to the peoples of the Arctic?
- what are the implications for the subsistence lifestyle of indigenous and other residents in the Arctic and how in turn will this impact their cultural and societal structures?
- what are "southern" governments' and scientists' responsibilities towards peoples living in the Arctic?
- what are the ethical parameters of the problem? do we need a new ethics?
- is there a need for an intergovernmental or international agency of the 8 Arctic countries to co-ordinate these efforts?
- do the 8 Arctic national governments (and/or the G-7 countries) have to earmark special aid research as well as funding?
- how can we manage to deal both with the past legacy, the present and future situations?

These are some of the questions that we as scientists should get the policy-makers to ask and the search for answers to these questions should be put of our recommendations.

We also have to work more closely with the peoples of the Arctic. Indigenous and non-indigenous politicians who live in the Arctic and who risk being directly affected by the radioactive waste there, tend to have a much more clear-cut view of the matter. They might well ask many of the same questions and will want straight answers. I would also suggest (and I say suggest because it could be presenptuous of me to speak for the peoples of the Arctic) that they will take - and indeed several indigenous groups have already done so - a strong stand

against any future use or dumping of radioactive material in the Arctic - they want a safe, clean and healthy environment and they will want whatever waste is already there cleaned-up or contained.

I would also suggest that the peoples of the Arctic might well use every available national and international means to achieve it. Scientists and governments will have to find answers to their questions, will have to work with the indigenous and non-indigenous peoples of the Arctic to achieve an environment free of radioactive waste and dangers.

With the changing political reality of the Arctic where more and more land titles, compensation money and political authority is turned over to the peoples of the Arctic, scientists and governments will have to take their perspective and their interests into account and our recommendations should clearly reflect this reality. I suggest to you that we undertake a concerted and dynamic international effort to deal with past, present and future radioactive waste and dumping in this new Arctic, and that we fully include the peoples of the Arctic in our work.

**LEGAL REGULATION OF ENVIRONMENTAL IMPROVEMENT OF THE  
RUSSIAN FEDERATION NUCLEAR COMPLEX ACTIVITIES  
IN ORDER TO PREVENT RADIOACTIVE POLLUTION  
OF ARCTIC SEAS**

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The question of environmental improvement of nuclear activities in the RF connected with prevention of radioactive pollution of the Arctic Ocean seas is of vital importance. The environment of the northern areas of the former USSR—and now Russia, and the bordering Arctic seas has to the highest extent suffered and is still suffering from the anthropogenic effect of radioactivity (except Chernobyl, of course). A large number of nuclear energy and nuclear-arms enterprises, marine bases are connected in the Russian Arctic area, and their activities lead to radioactive pollution of the environment. The situation is aggravated by hushing up of ecological data by the FSU authorities and the absence of adequate monitoring. It is enough to give just several representative examples that illustrate the radioactive situation in the Northern Russian territories and the bordering Arctic seas.

The original cause of the radioactive environmental pollution is the absence in the former USSR—and to some extent it is also true for Russia—of a safety concept of the nuclear complex operation, especially concerning radioactive wastes.

In the Urals, as the result of liquid radioactive waste dumping by "Mayak" nuclear industrial complex (Kyshtym, Chelyabinsk region), it lead to radioactive pollution of the river system, including Techa, Inset, and Tobol rivers flowing one into another.

A lake of polluted underground water has formed not far from Kyshtym complex, 30 km<sup>2</sup> in area and 4 mln m<sup>3</sup> in volume, which expands with the speed of 80 m per year. The polluted water partially flows into the cascade of reservoirs created along Techa-river for reducing the after-effects of its pollution by radioactive dumping, and there exists a real danger that the radioactive water could break into the open hydrographic system and pollute the Ob river-basin and the Arctic Ocean. Besides, this cascade of reservoirs has today become a potential source of pollution of the hydrographic system of Ob river-basin.

There are at least two other major northern enterprises of the same type in Russia—Tomsk-7 radiochemical plants (Sibchem-plant) and Krasnoyarsk-26 enterprise—with practically no available ecological information on their activities. The question of the unfavorable radioactive situation around Tomsk-7 plant was first brought up only in 1991 in connection with a possible threat to the environment of Tomsk region and Ob river-basin including the Arctic Ocean because of the same problem of dumping radioactive wastes. Radioactive waste disposal sites of Tomsk-7 plant are situated 10–12 km from Tom-river in porous soils, the concentration of radioactive sprays in the region considerably exceeds the maximum allowable level, and the content of radionuclides in the waste-waters on the border of the sanitary-protective zone (sodium-24) is only by the known categories three times higher than the allowable limit. Another accident leading to radioactive emission happened on this plant about a month ago. Radioactive consequences of the previous incidents and their after-effects on the environment still exist. There are grounds to suppose that the ecological situation around Krasnoyarsk-26 plant is also alarming.

No less alarming is the radioactive situation resulting from many years of activities of the USSR Navy in the Arctic. From 1959 to 1992 a total amount of about 24000 curies of liquid radioactive waste was dumped into the Arctic seas from the USSR Navy ships, and a total amount of over 15500 curies of solid radioactive waste were buried on land. There are 7 nuclear reactors loaded with nuclear fuel sunken in the gulfs along Novaya Zemlia shores and in the Novaya Semlia cavity in the Kara Sea. According to experts, the total level of radioactivity of the dumped objects is evaluated as 2.3 mln curies. In the Arctic

seas of the former USSR there were also dumped 10 used nuclear reactors without fuel with different levels of so-called pointed radioactivity reaching, depending on the maintenance conditions and the elementary composition of the reactor, 100,000 curies.

Radioactive pollution of the Arctic seas' environment also results from the emergency condition of the on-land storage plants on the Arctic Navy bases, from major accidents on nuclear submarines caused by destruction or fires of the reactors.

Thus, because of the violation of the cooling technology, one of the concrete containers in the storage of worked-out irradiated heating elements of a submarine reactor situated on shore of Andreyev Bay (Kola Peninsula) gave a leak, the highly-radioactive water getting into Liza-Fiord and Motov Bay. "Lepse", a similar technical sea-base, is also in an emergency condition, with the radioactivity level of the used heat-irradiating assemblies on board evaluated at about 750,000 curies. And at last, the after-effects of the accident on the Soviet submarine that sank on April 8, 1989 in the Norwegian Sea are really disastrous. The main danger comes from highly-rich nuclear fuel, three tons of it (about 1.5 t of enriched uranium in each reactor) being buried on the bottom of the Norwegian Sea under a busy sea-way, in the zone of intensive fishery. No less dangerous is the main component of warheads sunken together with the submarine-plutonium, a highly radio-toxic element with a long semi-decay period. The sea environment in the place of the accident already shows traces of radioactive isotopes which are washed out by the underwater streams from corroded nuclear topedos and the submarine.

Gradual realization of the need to work out an overall concept of safety for maintaining nuclear energy and nuclear arms enterprises, and other radioactive sources that can become a threat to people's life and health, the environmental situation has served as a starting point for forming legal basis of environmental regulation aimed at the reduction of the gap between the environmental safety provision in the nuclear-industrial and arm-building complex of Russia and other federal environmental programs. The

recently adopted Law on the Environmental Protection especially stipulates procedures and mechanisms of setting standards of the maximum allowable level of irradiation (article 29); the maximum allowable level of "strain" on the environment (article 33); ecological requirements are determined during the maintenance of nuclear energy objects (article 48/3) and while using radioactive materials (article 50). The Law imperatively bans the dumping of radioactive wastes and materials. What is especially important that all the mentioned environmental requirements set by the Law for all legal subjects are according to Article 55 of the Law "fully apply to military and defence objects, to military activities on the Russian Federation territory.

Radioactive safety aspects of the environmental protection were further developed in the "Statute on the Russian Federal Service of nuclear and radioactive safety control," in the RF President's Decree "On the organizations maintaining nuclear stations in the RF," in the RF President's Decree "On safety measures in the maintenance of nuclear energy and nuclear-arms complex enterprises, and radioactive sources threatening life, health and well-being of the population and the environment," and several other legal documents.

The major steps in the process of forming environmental legislation in the field of protection of nature from radioactive pollution will be two fundamental laws that are being discussed now by the Supreme Soviet of the RF. I am talking about two draft laws named "On state policy in the sphere of dealing with radioactive wastes" and "On usage of nuclear energy." Both drafts are based on the new for Russian law mentality conception of creating the legal regime of radiation safety for both human beings and environment during collecting, transporting, processing, storing and disposing of radioactive wastes and during use of nuclear energy in national economy and nuclear-arms enterprises of Russia.

In the draft "On state policy in the sphere of dealing with radioactive wastes" it is presumed that the main principle of the state policy of the RF here is the priority of protection of environment from

deleterious influence during dealing with radioactive wastes. Later on the whole draft is based on the similar approach. There is defined the key idea of safety while dealing with radioactive wastes, that in the first place includes prevention of pollution of environment while radioactive wastes are stored and/or disposed. There are introduced the main principles of state technical policy in the field of dealing with radioactive wastes, that give status of all obligatory laws to the rules and norms that before were only partly reflected in departmental legal acts. In particular, it lets us hope that the problem of over-accumulation of radioactive wastes in environment will be gradually solved. In the draft the legal status, order and conditions of work of special enterprises and organizations maintaining radioactive wastes is defined. That legal unification is necessary for the forming of a united and ecologically safe regime of dealing with radioactive wastes being in production cycle.

The draft "On usage of nuclear energy" is less worked out. The above-mentioned act sets forth the legal foundation and principles of safe usage of nuclear energy in any field, as well as that of life, health, and individual property protection, environmental protection from the deleterious effects of ionizing radiation. It is the first time that rights, responsibilities and obligations of state, state agencies, state officials and individuals in this sphere are stipulated.

Certainly the above-mentioned legal acts don't entirely solve the problem of preventing the radioactive pollution of environment. And what is more, in reality in Russia momentary departmental interests often prevail. As a result of it there are decisions of different governmental bodies and officials that contradict some laws and these decisions bear potential danger to environment and sometimes lead to ecological damage. The striking example of such decisions can be the Statute, approved by the Russian government, concerning building of nuclear stations. This statute contradicts Law on Environmental protection practically in all issues. Also, it can be easily forecasted that illegal activity of RF Arctic Navy concerning dumping of liquid and solid radioactive wastes will take place until 1995. Nevertheless, and I am sure of it, secret dumping of nuclear reactors with radioactive fuel and similar activities will not take



place in Russia anymore because of gradually growing state law mentality and understanding of our responsibility in front of the law. That's why the creating of the legal basis for the ecologically safe regime of work of the whole Russian nuclear complex lets us hope for the improving of the situation in the field of radioactive pollution of Arctic seas.

## Appendix I

Pre-conference material:  
1992 Planning Meeting Attendees  
Planning Committee Membership  
A Call for Abstracts

July 14, 1992

**LIST OF INVITEES/PARTICIPANTS  
PLANNING MEETING**

**RADIOACTIVITY IN THE OCEAN;  
COLD WAR LEGACY AND ENVIRONMENTAL SECURITY**

**July 14-16, 1992  
Woods Hole Oceanographic Institution  
Woods Hole, MA 02543**

**CANADA**

Mr. John Lamb  
Executive Director  
The Arms Control Centre

Dr. Joe MacInnis  
President  
Undersea Research Inc.

**RUSSIA**

Dr. V.P. Malyshev  
Scientific Advisor to the President on Radio-ecology  
Academician  
Russian Academy of Sciences

Prof. V.F. Menshikov  
Member, Russian Parliament  
Deputy Chairman, Committee on Environment

Dr. B.V. Nekipelov  
Scientific Advisor to the Minister of Nuclear Energy of Russia

Prof. A.N. Protsenko  
Head of the Department  
Institute of Nuclear Safety  
Russian Academy of Sciences

**UNITED STATES**

Dr. James M. Broadus  
Associate Scientist  
Director, Marine Policy Center  
Woods Hole Oceanographic Institution

Dr. Charles D. Hollister  
Senior Scientist, Department of Geology and Geophysics  
Vice President of the Corporation  
Woods Hole Oceanographic Institution

Dr. Hugh D. Livingston  
Senior Research Specialist  
Department of Marine Chemistry and Geochemistry  
Woods Hole Oceanographic Institution

Mr. Charles E. Meyers  
Section Head for the Arctic  
Division of Polar Programs  
National Science Foundation

Dr. Edward P. Myers  
Physical Scientist/Oceanographer  
National Oceanic and Atmospheric Administration

Mr. Peter B. Myers  
Director  
Board on Radioactive Waste Management  
U.S. National Academy of Sciences

Dr. Charles Newstead  
Office of Nuclear Technology and Safeguards  
Bureau of Oceans, Environment and Science  
Department of State

Mr. George Panteleyev  
MIT/WHOI Joint Program Student in Chemical Oceanography

Dr. Fred L. Sayles  
Senior Scientist  
Department of Marine Chemistry and Geochemistry  
Woods Hole Oceanographic Institution

Dr. Raphael Vartanov  
Senior Research Fellow  
Marine Policy and Ocean Management Program  
Marine Policy Center  
Woods Hole Oceanographic Institution  
(on leave from:  
Head, Section on the Environment and Ocean Development  
Institute of World Economy and International Relations  
Russian Academy of Sciences



### Conference Planning Committee

James M. Broadus, Woods Hole Oceanographic Institution  
Chris Garrett, University of Victoria, Canada  
Yuri Kuznetsov, Radium Institute\*, Russia  
John Lamb, Centre for Global Security, Canada  
Hugh D. Livingston, Woods Hole Oceanographic Institution  
V.P. Malyshev, Russian Academy of Sciences  
Valeriy F Menshikov, Russian Parliament  
Willy Ostreng, Fridtjof Nansen Institute, Norway  
Alexander N. Protsenko, Institute of Nuclear Safety\*, Russia  
Fred L. Sayles, Woods Hole Oceanographic Institution  
John N. Smith, Bedford Institute of Oceanography, Canada  
Raphael Vartanov, Institute of World Economy & International Relations\*,  
Russia

\*Russian Academy of Sciences





# A C A L L F O R A B S T R A C T S

## Radioactivity & Environmental Security in the Oceans

The purpose of this first conference is to re-evaluate, with a free and open exchange of scientific information, the current and future potential impact of artificial radioactivity in the marine environment.

Conferees will:

- assess current scientific knowledge and policy priorities
- recommend new research strategies for the Arctic and North Atlantic

The ending of the Cold War has created both an opportunity and a compelling need for a cooperative examination of these issues.

The conference will be convened by:

- Woods Hole Oceanographic Institution (USA)
- Canadian Centre for Global Security (Canada)
- Russian Academy of Sciences (Russia)
- Fridtjof Nansen Institute (Norway)

Abstracts should focus on one of the following four main topics related to artificial radioactivity in the Arctic and North Atlantic:

### I. Inventory

- Sources of materials
- Site studies: dumping
- Case studies: accidents
- Existing levels

### II. Routes, Rates & Reactions

- Transport by ocean currents, particles, & ice
- Data collection and assessment
- Biological (animal) and biomedical (human) effects
- Mobility in sediments

### III. Assessment and Remediation

- Assessment of sites of radioactivity
- Criteria for assessment
- Condition and containment of radioactivity
- Site radioecology
- Comparison to other hazards
- Site clean-up/remediation

### IV. Legal, Economic & Policy Priorities

- Responsibility, liability and ethics
- International and national governance and regulators
- Local population interests and critical groups
- Understanding and managing risk
- Economic choices

Abstracts can be no longer than one page. Not all abstracts can be accommodated. Abstracts will be selected based on quality, the extent of *new* information and perspectives, as well as relevance to the conference topics.

The deadline  
for submitting abstracts is  
February 15th.

Abstracts of selected abstracts will be mailed by March 15th.

Send abstracts to:

Hugh Livingston,  
Chairman, Conference Planning Committee  
Woods Hole Oceanographic Institution  
Woods Hole, MA 02543 USA

Telephone: +1-508-457-2000 extension 245

Fax: +1-508-457-2193

Telex: 7400462 HUGH UC

Omnet: H. Livingston

Internet: hlvivngston@whoi.edu

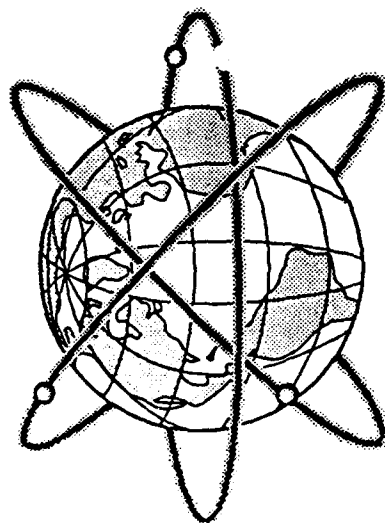
Participants will be asked to present an oral/poster session at the conference, and to bring their paper for later inclusion in the *Conference Proceedings*, which we plan to publish and distribute as quickly as possible. We also intend to report on the principal findings of the conference in the appropriate literature.

Limited funds may be available to defray costs.

NEW RESEARCH AND POLICY PRIORITIES IN THE ARCTIC AND NORTH ATLANTIC



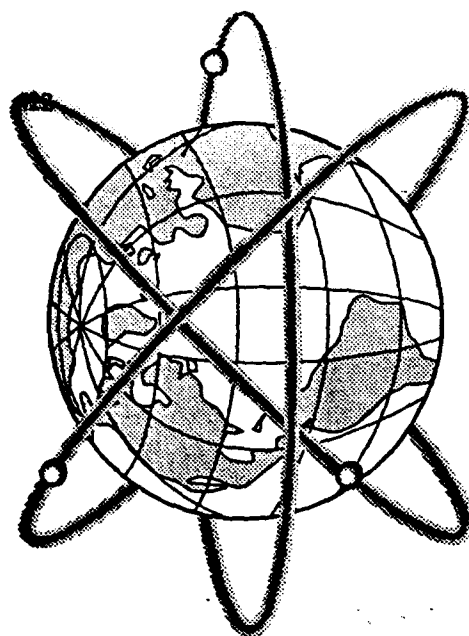
Woods Hole Oceanographic Institution  
Woods Hole, MA 02543 USA



#### Conference Planning Committee

- James M. Broadus, WHOI
- Chris Garrett, University of Victoria, Canada
- Yuri Kuznetsov, Radium Institute
- John Lamb, Centre for Global Security, Canada
- Hugh D. Livingston, WHOI
- V.P. Malyshev, Russian Academy of Sciences
- Valeriy F. Menshikov, Russian Parliament
- Willy Ostreng, Fridtjof Nansen Institute, Norway
- Alexander N. Protsenko, Institute of Nuclear Safety
- Fred L. Sayles, WHOI
- John N. Smith, Bedford Institute of Oceanography, Canada
- Raphael Vartanov, Institute of World Economy & International Relations

Russian Academy of Sciences



#### A CALL FOR ABSTRACTS

Due February 15, 1993

The First International Conference

## Radioactivity & Environmental Security in the Oceans:

New Research  
and Policy Priorities  
in the Arctic and North Atlantic

June 7 - 9, 1993

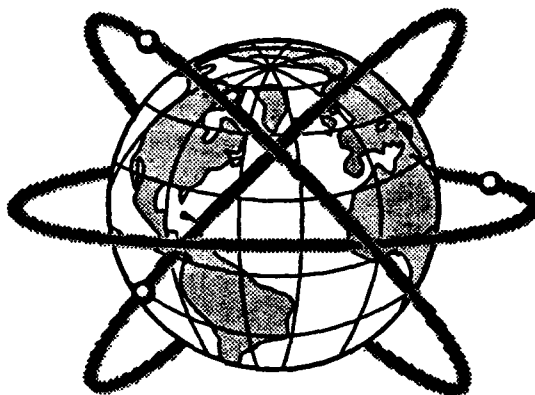
at the

Woods Hole Oceanographic Institution  
Woods Hole, Massachusetts USA

**Appendix II**

**Conference Program**

# CONFERENCE



***Radioactivity and Environmental Security in the Oceans:  
New Research and Policy Priorities in the Arctic and North Atlantic***

***June 7-9, 1993***

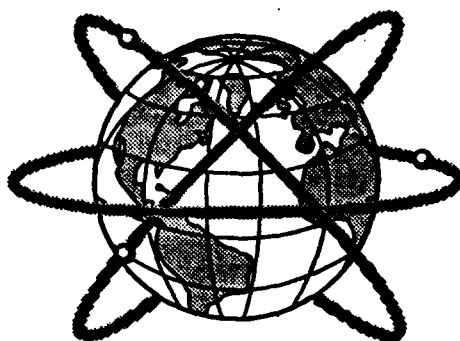
***Woods Hole Oceanographic Institution  
Woods Hole, Massachusetts USA***



***The purpose of this first conference is to re-evaluate, with a free and open exchange of scientific information, the current and future potential impact of artificial radioactivity in the marine environment.***

***Conferees will assess current scientific knowledge and policy priorities and recommend new research strategies for the Arctic and North Atlantic.***

***The ending of the Cold War has created both an opportunity and a compelling need for a cooperative examination of these issues.***




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***Conference Conveners and Planning Committee***

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James M. Broadus .....	Woods Hole Oceanographic Institution, USA
Christopher Garrett .....	University of Victoria, Canada
Yuri Kuznetsov .....	Radium Institute*, Russia
John Lamb .....	Canadian Centre for Global Security, Canada
Hugh D. Livingston .....	Woods Hole Oceanographic Institution, USA
V.P. Malyshev .....	Russian Academy of Sciences, Russia
Valeriy F. Menshikov .....	Russian Parliament, Russia
Willy Ostreng .....	Fridtjof Nansen Institute, Norway
Alexander N. Protsenko .....	Institute of Nuclear Safety*, Russia
Frederick L. Sayles .....	Woods Hole Oceanographic Institution, USA
John N. Smith .....	Bedford Institute of Oceanography, Canada
Raphael V. Vartanov .....	Institute of World Economy & International Relations*, Russia

***\*Russian Academy of Sciences***

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***Simultaneous translation and interpretation by Elena Mikhailova, Russia.***

***Translation and interpretation by Lesley MacLean, U.K.***

***Special thanks go to Sony Corporation of America and their Vice President, Craig C. Taylor, for their generous loan of the simultaneous translation equipment.***

*All sessions to be held in Clark  
507 unless otherwise noted*

### Sunday 6 June

4:00-7:00 pm Registration and  
poster assembly

5:30-7:00 pm Refreshments

### Monday 7 June

8:00 am Registration

8:30 am Opening remarks

Craig E. Dorman  
Director, Woods Hole Oceanographic  
Institution

8:45 am Organization and  
objectives of the conference

Charles D. Hollister  
Vice President of the Corporation,  
Woods Hole Oceanographic Institution

#### Session 1: Inventory

Co-Chairs: Yuriy A. Izrael, Director,  
Institute of Global Climate and  
Ecology, Russia, and Hugh D.  
Livingston, Woods Hole Oceanographic  
Institution

#### Introductory talks:

9:00 am Alexey Yablokov  
State Counselor for the Russian  
Federation on Public Health  
(presented by Vladimir Yakimets,  
Institute for Systems Analysis, Russia)  
*Arctic radioactive contamination from  
Russia - past and future*

9:30 am Poster Introductions:  
(Poster presenters only are listed. Full  
authorship will appear in conference  
proceedings.)

• Vitaliy Adushkin  
Institute for Dynamics of  
Geospheres, Russia  
*Novaya Zemlya nuclear test site  
and the Arctic Ocean radioactive  
pollution problem*

• M. Baskaran  
Texas A&M University, USA

*<sup>210</sup>Pb and <sup>137</sup>Cs-derived chronology  
and their fluxes in the Chukchi sea  
and Kotzebue Sound of the  
continental shelf regions of the  
Alaskan Arctic*

• Kathleen Crane  
Environmental Defense Fund/  
Hunter College/Lamont Doherty  
Earth Observatory, USA  
*Distribution of pollutants in the  
Arctic marine environment*

• Henning Dahlgard  
Risø National Laboratory, Denmark  
*Anthropogenic radioactivity in the  
Arctic Seas: time trends and present  
levels*

• Gennady A. Ivanov  
VNIIOkeangeologia, Russia  
*Radioecology of the Barents Sea and  
Kara Sea: Level of apprehension  
and research strategy*

• Peter Kershaw  
Ministry of Agriculture, Food and  
Fisheries - Fisheries Research  
Laboratory, UK  
*Radionuclides in the western  
Barents Sea: contribution of low  
level waste discharges from  
Sellafield, UK*

• Matthew Monetti  
United States Department of Energy  
*Three decades of <sup>90</sup>Sr deposition  
measurements in the northern  
latitudes*

• Mark Mount  
University of California, Lawrence  
Livermore National Laboratories  
*The estimated inventory of  
radionuclides in former Soviet  
Union naval reactors dumped in the  
Kara Sea*

• Gennady A. Nezhdanov  
Kurchatov Institute of Nuclear  
Energy, Russia  
*<sup>137</sup>Cs contamination of seawater  
around the "Komsomolets" nuclear  
submarine*

• Donald M. Schell  
Water Research Center, University  
of Alaska, Fairbanks  
*Bomb radiocarbon in Arctic  
Alaskan aquatic and terrestrial  
biota*

• Grant M. Raisbeck  
Centre de Spectrometrie Nucleaire  
et de Spectrometrie de Masse,  
France

*Anthropogenic <sup>130</sup>I in the North  
Atlantic*

• Yuriy Sivintsev  
Kurchatov Institute of Nuclear  
Energy, Russia  
*Radioecological situation on  
archipelago Novaya Zemlya (New  
Terra)*

• John N. Smith  
Bedford Institute of Oceanography,  
Canada  
*Radioactivity levels in Barents Sea  
sediments off Novaya Zemlya*

• Sergei M. Vakhlovsky  
Scientific Production Association  
"Typhoon", Obninsk  
*Radioactive contamination of the  
Barents and Kara Seas*

11:00 am Poster Session:  
informal discussions

12:30 pm Plenary Session:  
general discussion of posters  
and additional issues

1:00 pm Lunch

#### Session 2: Routes, Rates and Reactions

Co-Chairs: Leonid A. Savostin,  
Director, Shirshov Institute of  
Oceanology, Russia, and Charles D.  
Hollister, Woods Hole Oceanographic  
Institution

#### Introductory Talks

2:00 pm Yuriy A. Izrael  
Director, Institute of Global Climate  
and Ecology, Russia  
*Radioactivity and environmental  
problems for the seas and ocean*

2:30 pm Peter Schlosser  
Lamont Doherty Earth Observatory,  
USA  
*Large-scale circulation patterns and  
distribution of tritium and <sup>14</sup>C in the  
Arctic Ocean*

**3:00 pm Poster Introductions:**  
(Poster presenters only are listed. Full authorship will appear in conference proceedings.)

- Tengiz N. Borisov  
State Committee on Special Underwater Operations, Russia  
*Means of temporary localization of possible radioactive sources of pollution on sea and ocean bottoms*
- Warren Denner  
Environmental Research Associates, USA  
*Artificial radioactivity, circulation and mixing in the central Arctic*
- Lars Føyn  
Institute of Marine Research, Ministry of Fisheries, Norway  
*Radioactivity in the Barents Sea, past and present status, and its impact on fisheries*
- Igor L. Khodakovsky  
Vernadsky Institute of Geochemistry and Analytical Chemistry, Russia  
*Behavior of artificial radionuclides in seawater*
- Hartmut Nies  
German Federal Maritime and Hydrographic Agency  
*Temporal evolution of artificial radioactivity in the North and Baltic Seas after the reduction of the Sellafield discharges and the accident at Chernobyl*
- Nikolay A. Nossov  
Central Design Bureau for Marine Engineering "Rubin", Russia  
*Arrangement and technical facilities of radiation monitoring of the sunken objects with radioactive components*
- Vladimir K. Pavlov  
Arctic and Antarctic Research Institute, Russia  
*Peculiarities of the formation of structure and modification of hydrometeorological processes in the Kara Sea water area*
- Stephanie Pfirman  
Barnard College/Columbia University, USA  
*Potential transport of radioactivity in sea ice and the Upper Arctic Ocean*

- Marilyn R. Buchholtz ten Brink  
United States Geological Survey  
*Mobility of radioisotopes in marine surface sediments*
- Nikolay N. Vorontsov  
Institute of Developmental Biology, Russia  
*Destruction of the ecological chain in the Barents Sea, as a result of atmospheric nuclear explosions on Novaya Zemlya*
- A.J. Williams 3rd  
Woods Hole Oceanographic Institution, USA  
*Benthic flow forcing on the shelf by surface and internal waves*

**4:00 pm Poster Session:**  
Informal discussion

**5:30 pm Plenary Session:**  
general discussion of posters and additional issues

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## Tuesday 8 June

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### Session 3: Assessment and Remediation

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Co-Chairs: Kirsti-Liisa Sjöebloom, International Atomic Energy Agency, Vienna, Austria, and Frederick L. Sayles, Woods Hole Oceanographic Institution

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### Introductory talks:

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**9:00 am** Murdoch Baxter  
Director, Marine Environmental Laboratory, International Atomic Energy Agency, Monaco  
*Non-local radiological consequences of nuclear waste dumping in the Arctic Seas: a preliminary assessment*

**9:30 am** William L. Templeton  
Battelle, Pacific Northwest Laboratories, USA  
*Radiological assessments applied to dumping of radioactive wastes in the oceans*

**10:00 am Poster Introductions:**  
(Poster presenters only are listed. Full authorship will appear in conference proceedings.)

- Bill Curtis  
Environmental Protection Agency, USA  
*Application of U.S. Interagency coastal and marine monitoring programs to determining levels of radioactivity resulting from disposal of radioactive waste in Arctic Seas*
- Vitaly A. Eremenko  
International Center of Educational Systems, Russia  
*Treatment of data on sea pollution and dumping sites with the radioecological indices method*
- Timothy Francis  
Texas A&M University, USA  
*The application of offshore drilling technology to the entombment of artificial sources of radioactivity on the seafloor*
- Yuriy V. Kuznetsov  
V.G. Khlopin Radium Institute  
*Radioactive contamination of Northern Seas: approaches to the assessment of impact on the marine environment and man*
- Vitali N. Lytsaov  
Ministry for Environmental Protection, Russia  
*Radioecological risk assessment for solid radioactive waste dumped into the Kara sea by the former USSR*
- Steven Mudge  
School of Ocean Sciences, Gwynedd, Wales, UK  
*Radiological assessment of the Ribble Estuary*
- N. James Parks  
University of California, Davis, USA  
*Evaluation of liquid and emulsion scintillation spectrometry for quantitating alpha and beta emitters in seawater and other matrices*
- Alexander N. Protsenko  
Nuclear Safety Institute, Russia  
*Problems of evaluation and management risk of radioactive ocean contamination*
- Vladimir Samoilov  
Central Design Bureau for Marine Engineering "Rubin", Russia

*Plans for a 1993 cruise to the Komsomolets site*

- Kirsti-Liisa Sjoebloom  
International Atomic Energy  
Agency, Vienna, Austria  
*International Arctic Seas Assessment Project (IASAP)*
- David L. Stein  
National Undersea Research  
Program, National Oceanic and  
Atmospheric Administration, USA  
*Assessment of deep-sea nekton and  
their role in radionuclide  
transfer*
- Per Strand  
Norwegian Radiation Protection  
Authority, Norway  
*Plans for a Russian/Norwegian  
cruise to the Kara Sea*
- Thomas H. Suchanek  
University of California, Davis, USA  
*Radionuclide burdens in commercial  
deep-sea fishes and intertidal  
mussels from the vicinity of the  
Farallon Islands nuclear waste  
dump site, California*
- Nathalie J. Valette-Silver  
National Oceanic and Atmospheric  
Administration, USA  
*Radionuclides in bivalves collected  
along the coastal U.S.: Results from  
the NOAA NS&T program*

11:00 am                      Poster Session:  
   Informal discussion

12:30 pm                      Plenary Session:  
   general discussion of posters  
   and additional issues

1:00 pm                                      Lunch

#### **Session 4: Legal, Economic and Policy Priorities**

Co-Chairs: Marianne Annelise  
Stenbaek, Director, Centre for  
Northern Studies and Research, McGill  
University, Canada, and Raphael  
Vartanov, Woods Hole Oceanographic  
Institution

#### **Introductory talks:**

2:00 pm                      Valeriy F. Menshikov  
Deputy Chairman of Parliamentary

Committee on the Environment, The  
Supreme Soviet, Russia

*Some aspects of the legal regulation  
of radioactive waste disposal at sea*

2:30 pm      Marianne Annelise Stenbaek  
Director, Centre for Northern  
Studies and Research, McGill  
University, Canada

3:00 pm                      Brief Introductions:  
(Poster presenters only are listed. Full  
authorship will appear in conference  
proceedings.)

- Yuriy G. Baraegov  
Institute of World Economy and  
International Relations, Russia  
*Certain legal issues concerning  
prevention of land-based radioac-  
tive pollution of the Arctic marine  
environment*
- Giff Curtis  
Greenpeace, USA  
*International and national  
governance and regulators*
- Joshua Handler  
Greenpeace, USA  
*Data on nuclear submarine  
accidents in the US, Russian,  
French, British and Chinese navies*
- Lassi Heininen  
Arctic Centre, University of Lapland,  
Finland  
*Radioactivity from the military  
sources: A message to take new  
policy priorities*
- I.I. Kuz'min  
Kurchatov Institute of Nuclear  
Energy, Russia  
*The radioactive contamination of  
the North Polar basin: the risk of  
communication*  
*Global regional and national  
nuclear risk management: the use  
of ALARA in decision-making on the  
costly international cleanup effort  
of radioactive contamination in the  
shallow seas*
- Bruce Molnia  
United States Geological Survey  
*Interagency Arctic Research Policy  
Committee (IARPC) involvement in  
the international Arctic contamina-  
tion issue*

- Renat A. Perelet  
Institute of Systems Studies, Russia  
*Environmental security and  
governance issues in the Arctic  
Ocean*
- Michael Vekler  
Institute of World Economy and  
International Relations, Russia  
*Legal regulation of 'Ecologization' of  
the activities of the Russian  
Federation nuclear complex in  
order to prevent radioactive  
pollution of the Arctic Ocean*

4:00 pm                      Poster Session:  
   informal discussion

5:30 pm                      Plenary Session:  
   general discussion of posters  
   and additional issues

6:00 pm                                      Barbecue  
   at Pease House

### **Wednesday 9 June**

9:00 am                      Working Groups  
Co-Chairs: John Smith, Bedford  
Institute of Oceanography, Canada,  
Stephanie Pfirman, Barnard College,  
USA, Christopher Garrett, University of  
Victoria, Canada, and John Lamb,  
Centre for Global Security, Canada  
*These groups will be based on the  
session topics and are to discuss  
issues raised subsequent to these  
sessions and will consider questions  
and issues, e.g. future directions,  
which will be addressed in the final  
Plenary Session.*

12:30 pm                                      Lunch

1:30 pm                      Plenary Session  
Chair: Charles D. Hollister, Woods Hole  
Oceanographic Institution  
*Presentations of Working Group  
Discussions (Group Chairs)  
General Discussion  
Recommendations*

4:30 pm                      Closing Remarks:  
Charles D. Hollister, Woods Hole  
Oceanographic Institution

### **Thursday 10 June**

10am-Noon                      Press Briefing





## Appendix III

List of Registered Attendees  
with addresses

**Radioactivity and Environmental Security in the  
Oceans: New Research and Policy Priorities in the  
Arctic and North Atlantic  
June 7-9, 1993**

**Participants List (6/28/93)**

1 Vitaly Adushkin  
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Woods Hole, MA 02543  
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13 Geoff Bourne  
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